Assessment of the Aquatic Effects of Mining in Canada:

AQUAMIN

Supporting Document II: Western Canadian Shield

Case Studies: Volume 1
Nanisivik and Lupin Mines

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Supporting Document II: Appalachian Region

Case Studies: Volume 1
Nanisivik and Lupin Mines

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1 Lupin Mine

1.1 Summary of Mine Site Information

1.1.1 Site Setting

The Lupin Mine is located in the District of Mackenzie of the Northwest Territories (65° 46'N, 111° 14'W) on the northwest shore of Contwoyto Lake. The site is approximately 425 km NNE of Yellowknife and 300 km SE of Coppermine (Figure 1.1). Contwoyto Lake, with an approximate surface area of 95,900 ha and maximum length of 110.4 km, is the recipient of tailings decant from the mining operations, via Seep Creek, which flows into Inner Sun Bay (Figure 1.2).

The Lupin Mine site lies in the Bear-Slave Upland division of the Kazan Physiographic Region of Canada (BEAK 1980). The area surrounding the mine has low relief (elevation range - 470-505 m) and can be classified into low-lying areas (swamps and shallow lakes) that serve as drainage, gentle slopes, and flat uplands (BEAK 1980). Permafrost underlies the entire area to a depth of 370 m, with the active layer estimated to be 1 m during the summer thaw (BEAK 1980).

Figure 1.1: Location of the Lupin Mine

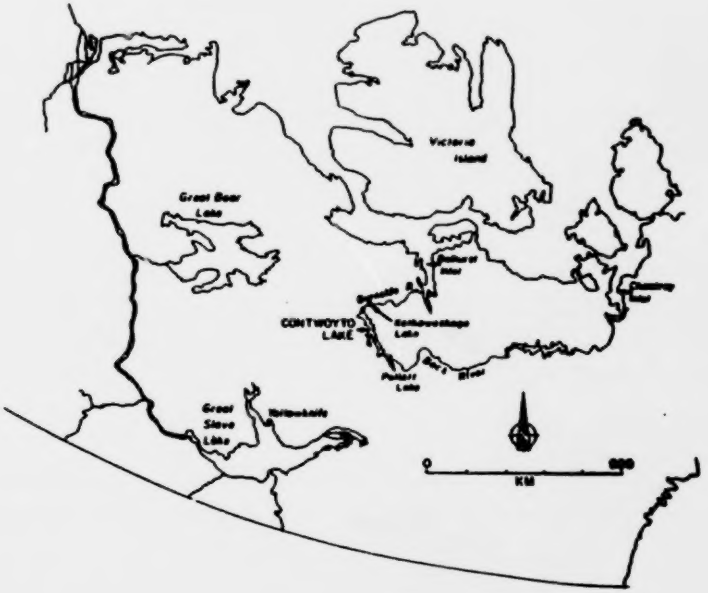
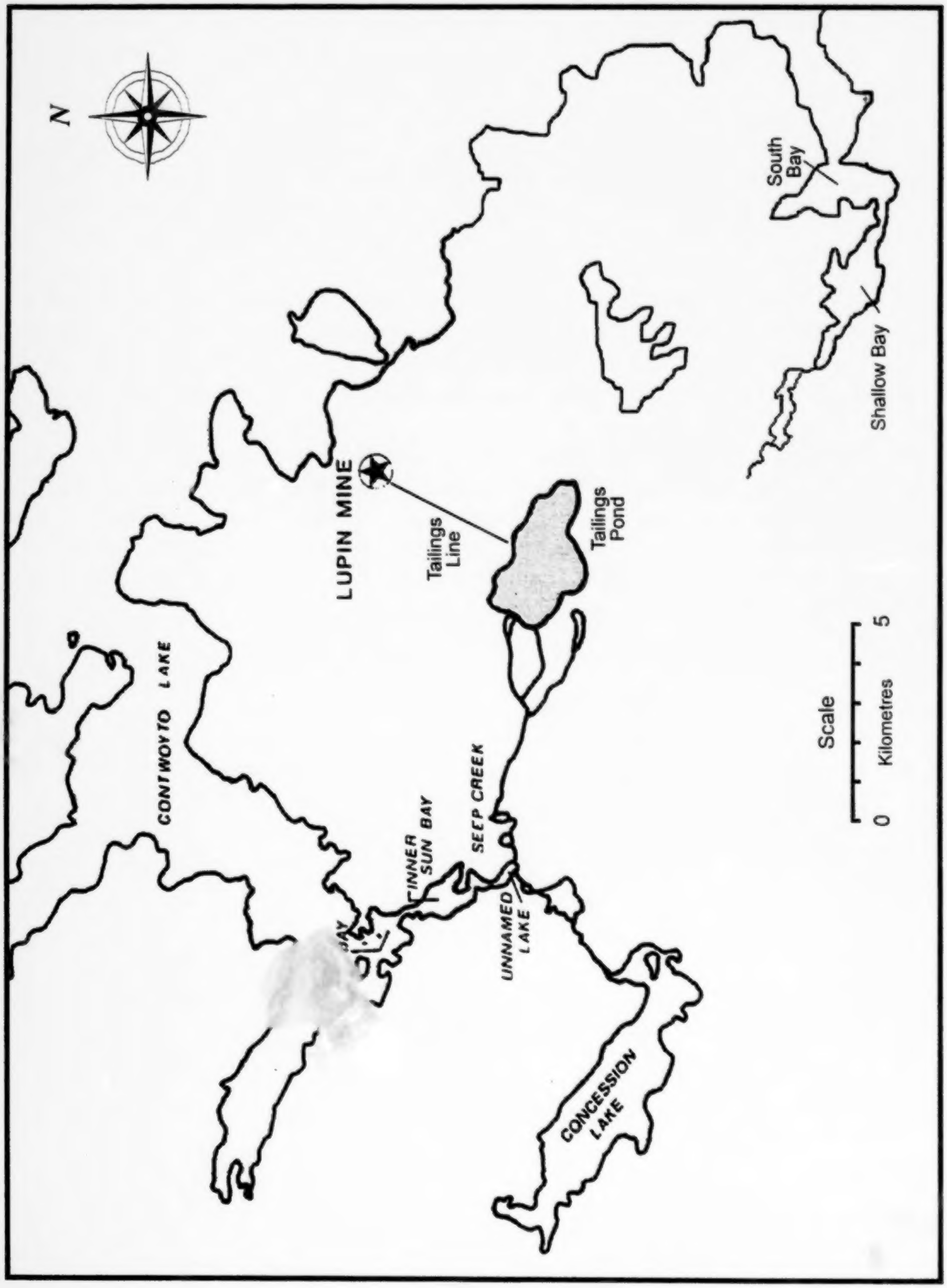

Victoria Island
Great Bear Lake
Bathurst Inlet
Chantrey Inlet
CONTWOYTO LAKE
Pellatt Lake
Back River
Great Slave Lake
Yellowknife
0
KM

Figure 1.2: Map of the Lupin Mine site


N
LUPIN MINE
Tailings Line
Tailings Pond
South Bay
Shallow Bay
CONTWOYTO LAKE
INNER SUN BAY
SEEP CREEK
UNNAMED LAKE
BAY
CONCESSION LAKE
Scale
0
5
Kilometres

Glaciation deposited a layer of glacial till (silty sand with gravel and boulders), which has been overlain in place by glacio-fluvial and glacio-lacustrine deposits of sand and gravel. In most low-lying areas, an organic layer generally less than 6 cm is present (BEAK 1980). The area in the immediate vicinity of the mine site is a combination of shallow lakes, streams, and swamps (Figure 1.2). The numerous drainage systems that surround the tailings ponds all eventually flow into Contwoyto Lake, but placement of dams around the tailings areas directs flow into Sun Bay.

1.1.2 Geology

Lupin Mine is located within the Slave Province of the Canadian Shield and bedrock is composed of metamorphic rock. The immediate area of the mine is part of the Yellowknife Subgroup and bedrock is composed of a mixture of metamorphosed argillite, siltstone, slate, greywacke, and quartzite. The ore occurs in veins of gold-bearing sulphides hosted in beds and lenses of gneiss (BEAK 1980).

Echo Bay Mines calculated the Lupin gold reserves in 1980 to be 1,852,000 tons of ore at a grade of 0.44 oz Au/ton of in-place reserves and 1,833,000 tons of 0.40 troy oz Au/ton ore of mineable reserves.

1.1.3 History

The Lupin Mine is owned and operated by Echo Bay Mines. Mine development began in 1980, with commercial production beginning in October, 1982. Using the Merrill-Crowe process for gold recovery, the initial mill production rate averaged 975 tons/day. The current production rate is approximately 1500 tons/day. Gold is refined to bullion in an on site furnace.

1.1.4 Effluent Treatment

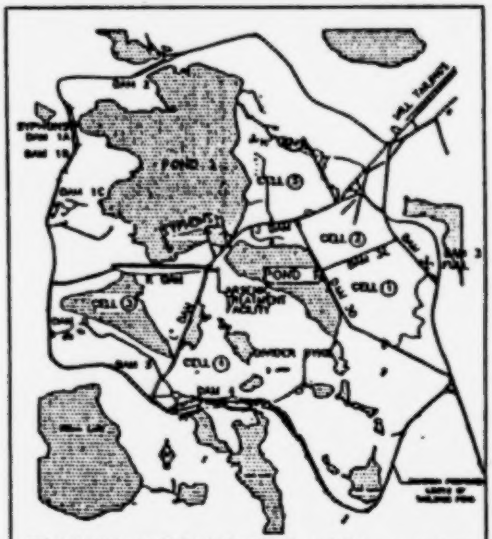
Approximately 50 % of the tailings generated from the operation are used as paste backfill underground. The remainder of tailings and all waste water from the operation are released to the tailings management area approximately 6 km SW of the mine site. Prior to release of tailings from the pre-aeration and cyanide filtering processes chlorine gas is bubbled through the tailings to reduce cyanide concentrations.

The tailings management area at the Lupin Mine, which is designed to optimize natural degradation processes, has evolved as the operation has expanded. The current area is illustrated in Figure 1.3. It includes three solids retention cells and two polishing ponds. The individual cells prevent intermixing of fresh tailings with water that has been in the impoundment and subjected to natural degradation for at least one year. Originally, all liquid wastes from the Lupin operation were to be contained in the tailings management area, with no effluent discharge (Echo Bay Mines Ltd. 1994). However, as the mill capacity was expanded the capacity of the tailings management area was exceeded, and it became necessary to discharge effluent. Discharge generally begins no sooner than July 15 and usually continues for a period of two weeks. In 1990, an estimated 837,000 m^3 of effluent was decanted between July 2 and July 16.

The tailings management system includes:

1) Tailings solids and waste water pumped to solids retention cells for primary settling of solids.
2) Solids retention cells operated in series in a sequential decant and fill manner, prior to decent of liquids to first polishing pond
3) Iron sulphate solution added prior to decant of clarified liquids from first polishing pond to second polishing pond. Iron sulphate controls arsenic concentrations in effluent by precipitating the arsenic[1]
5) Decant from the second polishing pond is discharges to Seep Creek.

Figure 1.3: Configuration of the tailings managment area in 1994



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[1] Note that effluent was not treated for arsenic during the 1985 discharge period.

1.1.5 Effluent Compliance Requirements

The Lupin Mine operates under a Water Licence issued by the Northwest Territories Water Board. The first licence was issued in 1981 and an amended licence was issued in 1985 when effluent discharge began. Effluent discharge limits in the current Water Licence are presented in Table 1.1.

Table 1.1: Effluent quality standards for discharge water from tailings containment area

Parameters	Maximum Average Concentration (mg/L)	Maximum Concentration of any Grab Sample (mg/L)
Total Arsenic	0.5	1.0
Total Cyanide	0.8	1.6
Total Copper	0.15	0.3
Total Lead	0.05	0.1
Total Nickel	0.1	0.2
Total Zinc	0.3	0.6
Suspended Solids	15	30

The maximum rate of effluent discharge from the Tailings Management Area is 60,000 m^3/day

1.1.6 Sewage Disposal from Lupin Mine

Sewage and domestic grey water is pumped to the Sewage Lake Disposal System for natural degradation (Figure 1.2). The lakes are located south of the mine and empty via a creek into Contwoyto Lake (Echo Bay Mines Ltd. 1994). These lakes also receive discharge from a small, inactive tailings disposal area which generates some acidic drainage.

Table 1.2: Effluent quality standards for discharge water from the Sewage Lakes Disposal System

Parameter	Maximum Average Concentration (mg/L)
Total Arsenic	0.05
Total Copper	0.20
Total Nickel	0.05
Total Zinc	0.50
BOD	30
Suspended solids	35
Faecal coliform	1,000 colony forming units/100 mL
pH	›6.0

1.1.7 Compliance Record

Compliance level with limits in the Water Licence has been very high. There have been no non-compliance events during the decant discharge periods, although some concern has been expressed by Echo Bay as the zinc and nickel levels that have approached the Maximum Average Concentration limits, specifically during the 1992 and 1993 decant periods.

All effluents discharged from the Sewage Lakes Disposal System have met or surpassed the Effluent Quality Standards as set out in the Water Licence, although some difficulty has been experienced in maintaining the effluent pH above 6.0 and this was achieved by the addition of lime in 1993 and 1994.

Spills and unauthorized discharges that have occurred during the existing Water Licence term have been limited to the following:

- miscellaneous oil and solvent spills during storage or delivery around the site
- minor accidental tailings spills from the emergency dump ponds and associated facilities (1993)
- a single major spill from the Tailings Management Area in 1992 due to deficiencies in estimated capacity of the "retaining structure". Echo Bay Mines Ltd. was charged under Section 36(3) of the Fisheries Act.

1.1.8 Acid Mine Drainage

Lupin tailings, given the proper conditions, will oxidize and produce byproducts necessary for the formation of acidic runoff. However, typical Lupin tailings contain approximately 3% sulphur (total) and have a NP/MPA (Neutralizing Potential/Maximum Potential Acidity) ratio of less than three which indicates a low potential for acid generation and no concern with respect to AMD. Since the licence renewal in 1990 Echo Bay Mines Ltd. has increased its research and development activity with regard to the acid generating potential of the tailings and mitigative/abandonment measures for eventual closing of the tailings management area.

1.2 Quantity and Quality of Data Available for the Site

1.2.1 Assessment of data quality

1.2.1.1 Water Quality

Two studies of water quality in the vicinity of the mine have been conducted. Reid Crowther (1985a) described seasonal and annual variability of water-borne metals and other constituents (pH, total dissolved solids, calcium, magnesium, sodium, sulphate) for the ten locations, including two locations in Contwoyto Lake (Inner Sun Bay, Shallow Bay). All of the receiving waters in the area were examined.

Mudroch and Sutherland (1988) described water quality at stations in Contwoyto Lake before and after the first effluent decant in 1985. They included stations in Inner Sun Bay, the channel leading to Outer Sun Bay and four stations in Outer Sun Bay, including a presumed control station. Samples were collected at both the surface and bottom of the water column. Some contamination of "blank" samples by copper and zinc was evident. After decant, metal levels increased at most stations, including the presumed control station. Changes in metal levels were tested statistically and depicted graphically. The data were useful for indicating the general flow pattern tendencies of the plume from the tailings pond in Contwoyto Lake.

It is important to note that no water quality data for Contwoyto Lake from after 1985 was available for the preparation of this case study. As a result, only limited conclusions can be reached regarding water quality in Contwoyto Lake since the 1985 effluent decant was not representative of later effluent in later years, particularly with respect to arsenic.

1.2.1.2 Sediment Quality

Reid Crowter (1985a) described sediment quality at eight locations including Inner Sun Bay and South Bay in Contwoyto Lake during 1982 and 1984. Methodology in 1982 involved measurement of constituents on whole sediment samples, whereas in 1983 and 1984 constituents were measured only in the less than 53 μm fraction, confounding the ability to interpret differences between data from 1982 and the following two years. Most of the potential receiving waters were surveyed as well as control areas that should not have been receiving mine-tailings runoff at that time.

Mudroch and Sutherland (1988) described sediment quality at stations in Contwoyto Lake before and after the first effluent decant in 1985. They included stations in Inner Sun Bay and four stations in Outer Sun Bay including a control station. They also collected cores to determine the vertical distribution of metals. Collection of these cores provides an additional measure for historical metal loadings at a specific site and removes some the within-location variability.

Porter *et al.* (1992) described sediment quality at six station including a new station. Sediment cores were similar to those processed during 1985, but only the top 1 cm was retained for analysis.

1.2.1.3 Benthic Invertebrates

Four distinct studies have documented benthic communities in Contwoyto Lake and surrounding areas.

Mine-Site Studies

Moore (1978) described benthos, plankton and algae from Contwoyto Lake and nine other lakes in the area. A basic description and list of benthic fauna in the lakes was established.

Reid-Crowther (1985a) surveyed locations on the mine site considered most susceptible to mine related effects plus two locations in Contwoyto Lake. Reid Crowther (1985a) suggested it was difficult to relate among-lake differences in communities to mining activities because habitats within each lake were too different. Further, the lack of proper reference lakes or streams (i.e., unexposed to mining activities) also made it impossible to relate changes in the benthic communities to changes water quality due to mining activities. The study design of Reid Crowther (1985a) could have been improved had they re-surveyed some of the same locations surveyed by Moore in 1975. The authors concluded that natural environmental variables accounted for much of the variation in benthic community structure, but little attempt was made to standardize for habitat variables in the sampling design. Consequently, the data collected by Reid Crowther (1985a) cannot be used to describe the extent or magnitude of effects on the mine site.

Contwoyto Lake Studies

In 1985, Mudroch and Sutherland (1988) collected benthos before (August) and after (September) the first effluent decant. Benthos were collected at five locations in Contwoyto Lake: one station in Inner Sun Bay, three stations at the mouth of Outer Sun Bay, and a fifth (control) station just beyond Outer Sun Bay. In 1990, Porter *et al.* (1992) surveyed benthos at the same stations as Mudroch and Sutherland (1988), as well as one additional site in the middle of Contwoyto Lake. That study was to document changes in the benthic community five years after the first decant.

In general, these studies used similar field methodologies for collection of samples, but lacked consistent laboratory protocols. Further, appropriate temporal or spatial reference stations were lacking. Inadequate preservation of samples collected in 1990 made it difficult to accurately identify snail and clam specimens (EVS Consultants 1991). In addition, sample washing procedures in 1990 may have resulted in more specimens being retained in the samples.

Porter *et al.* (1992) identified a much higher number of species than any of the other surveys (73 taxa in 1990 versus 32 in the previous survey). In part, this was due to the identification of copepods and ostracods which are often left unidentified. The difference was also due to the improved identification of aquatic mites, chironomids, oligochaetes and sphaeriids, and the presence of amphipods, miscellaneous diptera and trichoptera in the samples. Inconsistencies in taxonomy between the 1985 and 1990 surveys are listed in Table 1.3 and include differences in the species of sphaeriids (clams), gastropods (snails) and caddisflies (Trichoptera).

Finally, the stations which have been repeatedly sampled over the years in Contwoyto Lake cannot be used to succinctly demonstrate changes in the benthic community over time due to mine activities. All stations that have sampled before and after mining decant activities are exposed to mine effluent, included the control station in Outer Sun Bay, making it difficult or impossible to determine the magnitude and extent of mining related effects, and the causes of effects. In the absence of adequate control data it is possible to attempt to relate changes in the benthic community to metal-exposure level in the effluent plume, but this is not an ideal method.

Table 1.3: Inconsistencies in benthic taxonomy among surveys conducted on Contwoyto Lake

Group	Inconsistency
Shaeriidae	The 1990 survey found *Pisidium conventus, P. lilljeborgi* and *P. ventricosum*, but not *P. idahoense*. In the earlier surveys, the investigators may have confused *P. casertanum* with the other taxa
Gastropoda	The 1990 survey found only *Gyraulus deflectus*, while earlier surveys found only *Valvata sincera*. The taxonomist in 1990 indicated some question on the identity of the species (Kilgour, 1991)
Mites	The 1990 survey found 7 species whereas the other surveys found only 2 species
Chrironomids	The 1990 survey found 30 genera whereas the 1985 survey found only 2 species
Hydra	Was not reported until 1990
Trichoptera	A common limnephilid caddisfly found in 1983 at the Inner Sun Bay station (*Grensia*), may have been identified as *Apatania* in 1990
Oligochaetes	Not identified to species until 1990

1.2.1.4 Fisheries

Roberge *et al.* (1986) conducted an extensive study of the fish communities present in Contwoyto Lake during the summers of 1981 and 1982. Large numbers of lake trout (n = 525) were caught using gill nets and data were provided on length, weight, sex, age, and maturity. They also conducted a tagging study in which 161 lake trout were tagged in hopes of estimating population size by mark-recapture methods. This paper provides a good amount of descriptive baseline data. Some data, but from much smaller sample sizes, is available for arctic char, round whitefish and cisco.

A series of studies were conducted to determine the metal concentrations in muscle and liver tissue of fish from Contwoyto and Concession Lakes (Ash and Harbicht 1991; R.L. &L. Environmental 1989, 1994; Reid Crowther 1985b), both prior to the commencement of mining operations and after operations had begun. Although Roberge *et al.* (1986) sampled extensively in Contwoyto Lake, sampling in other studies was more restricted. Reid Crowther (1985b) initially obtained fish samples from three sites in Contwoyto Lake and a from a few smaller lakes. More recent studies (Ash and Harbicht 1991; R.L.&L. Environmental 1989, 1994) restricted sampling to Inner Sun Bay, one site in Contwoyto Lake and Concession Lake. Fish were caught using gill nets and muscle and liver tissues were analysed for arsenic, cadmium, copper, mercury, nickel, and zinc. These reports were thorough and include length, weight, age, and sex data in their appendices.

Sampling of fish tissues for metal residue analysis involved standard techniques to minimize the contamination of samples. Fish tissues residues analysis was conducted on muscle samples from each fish and on pooled samples of liver tissue. The sample sizes for muscle tissue were adequate to determine trends for metal levels, while the pooled samples for livers resulted in smaller sample sizes. Authors pooled metal levels from all sites for data correlation analysis, instead of conducting an initial site by site correlation. Correlations simply suggest whether a significant relationship exists between two variables. Comparisons of metal content of tissues between sites was conducted using one-way ANOVA. This method of comparison has some problems, since all the not-detected values are incorporated into the means as a value of half the detection limit. For comparison of mercury residues, a one-way ANOVA with comparing mean values does not take into account the relationship between body size and mercury content of tissues.

Reid Crowther (1984, 1985a,b) conducted major fisheries assessments during all four seasons of 1983 and 1984. Box traps were used in Decant and Seep Creeks to monitor upstream movement of fish. Fish were also collected using gill nets, backpack electrofisher, beach seine, angling, and set lines. A tagging program was also conducted with arctic grayling, arctic charr, and lake trout. Population sizes could not be estimated due to the small number of tag returns. Stomach content analysis was also conducted for fish that were caught. An attempt was also made to collect fish harvest data from mine employees by interviews and questionnaires. Habitat descriptions were also provided for stream areas that were sampled. Most of the data is descriptive in nature and few quantitative comparisons are made.

Moore (1978) provides cursory summaries of presence/absence data for fish with some mean lengths and weights. The emphasis of this report was on phytoplankton and zooplankton community structure. The data summaries are very general (even for water chemistry) and should be considered descriptive at best.

1.2.2 Adequacy of Monitoring Programs

None of the studies clearly stated an objective or testable hypothesis in the design of their monitoring programs. The choice of some of the sample locations is questionable. Proper control stations should have been established when it was determined that effluent decant would be necessary. For example, Contwoyto Lake station 2 is not an adequate reference site since water quality monitoring indicated that this site is exposed to effluent. For fish sampling the sampling stations included one exposed area (Inner Sun Bay), and two reference areas (Concession Lake and the main body of Contwoyto Lake). These reference areas are the best available but they are not ideal. Fish may move between Inner Sun Bay and the Contwoyto Lake reference site, and although Concession Lake is separated from

Contwoyto Lake, it is at least 10-fold smaller so fish populations may have different characteristics.

Lake trout may not be the ideal sentinel species for monitoring at Lupin due to their slow growth and reproduction rates and subsequent susceptibility to harvesting pressure. Further, they may be highly mobile within the Contwoyto Lake system. However, lake trout are an important species to the local aboriginal fishery.

1.3 Results of Environmental Monitoring Studies

1.3.1 Water and Sediment Quality

1.3.1.1 Mine-Site Studies

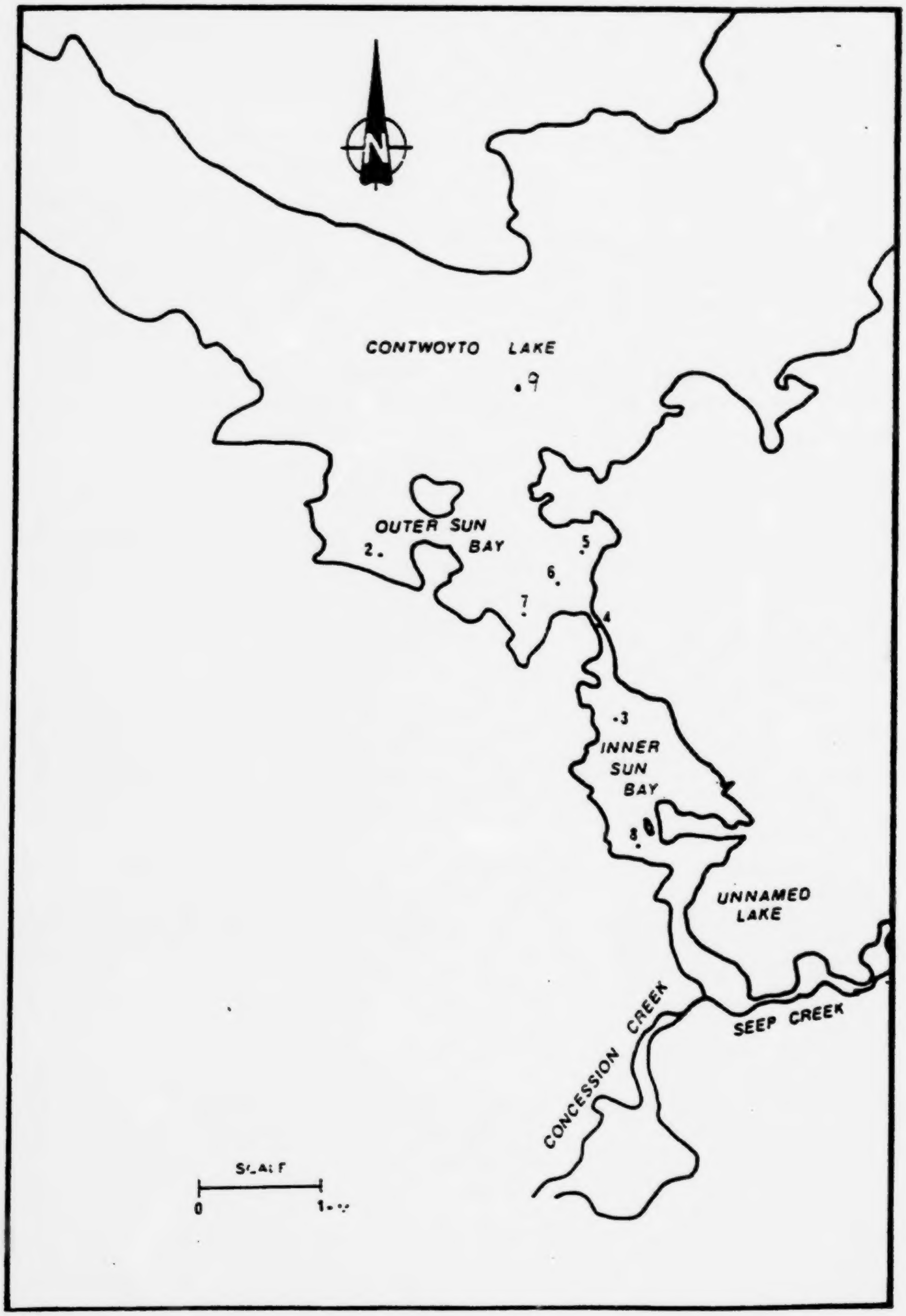
At the mine site changes in water and sediment quality parameters were negligible and difficult to attribute of mining activities. A small lake north of the tailings management area, referred to as Dam 2 Lake, consistently had the highest levels of copper, nickel and zinc in both water and sediments, but apparently has always had high levels of metals (Reid Crowther 1985a). During 1984, cyanide was found in water samples from Dam 2 Lake and Seep Creek, but was also found in control samples from Concession Creek. Metal and arsenic concentrations are elevated in Seep Creek, which receives the effluent decant. However, arsenic concentrations are also elevated in Concession Creek, suggesting that at least some of the arsenic is from natural sources.

In addition to increased waterborne metal levels in Seep Creek, effluent decant also influence flow rate and possibly water temperature in Seep Creek. Seep Creek is a seasonal stream, so the effect of the effluent decant of creek flow rate is variable. Some years, the effluent decant constitutes 100 % of the creek flow.

1.3.1.2 Contwoyto Lake Studies

Locations of water and sediment sampling stations in Contwoyto Lake are shown in Figure 1.4. Data from 1985, after the first effluent decant, indicate that the effluent plume extended at least as far as station 2, the intended control stations. In particular, arsenic concentrations were elevated at station 2 (Mudroch and Sutherland 1988). However, it is important to note that the 1985 effluent decant was a unique event, since this decant was not originally planned, and there was no treatment methods in place to reduce arsenic concentrations in effluent. Water quality data for Contwoyto Lake after 1985 were not available for the preparation of this case study.

Figure 1.4: Location of water, sediment and benthic invertebrate sampling stations in Contwoyto Lake


N
CONTWOYTO LAKE
9
OUTER SUN BAY
2
5
6
7
4
3
INNER SUN BAY
8
UNNAMED LAKE
CONCESSION CREEK
SEEP CREEK
SCALE
0
1 km

From Inner Sun Bay, the effluent plume moves in two directions depending on position in the water column. In the upper water column, the plume leaves Inner Sun Bay in the direction of Station 5. Bottom waters from Inner Sun Bay apparently move in the direction of station 6. For the 1985 data, the ranking of stations according to their exposure to effluent was 2 ‹ 7 ≤ 6 ‹ 3 = 4 ‹ Seep Creek.

In 1985, sediments at stations 6 and 7 had elevated levels of copper, zinc and nickel, while there were no significant increases in sediment-metal levels at station 2. Mudroch and Sutherland (1988) also reported changes in sediment type (a 16% increase in mean loss on ignition and a 13% increase in fine particles) at station 7 from August to September, 1985. Porter *et al.* (1992) added station 9 since it represented deep-water habitat. Mudroch and Sutherland (1988) hypothesized that the deeper parts of the lake were depositories for sediments and would have higher metal levels. In the core from station 9 there was no apparent trend in metal levels suggesting no recent change in metal levels at that site. Porter *et al.* (1992) reported various changes in metal levels in sediment compared with the results of previous sampling. For example, arsenic concentrations increased at stations 3 and 5 only, while copper and zinc increased at stations 2 and 5.

Metal concentrations in surface water in Seep Creek, Concession Creek and Inner Sun Bay from 1983 to 1983 are presented in Table 1.4. Metal concentrations in sediments from Sun Bay from 1982 to 1985 are presented in Table 1.5.

Table 1.4: Mean concentrations (mg/L) of waterborne elements in samples from Seep Creek, Concession Creek, and Inner Sun Bay (1983-1985)

Sampling Date	Element					
	Ca	Hg	As	Zn	Ni	Cu
			Seep Creek			
Aug. 1983	3240	1990	< 1	14	17	3.2
Sept. 1983	7110	2210	< 1	17	22	2.1
Aug. 1984	3700	2380	< 1	16	30	3.0
Sept. 1984	6480	4050	< 1	20	41	1.1
Aug. 1985	4570	2350	< 1	4.1	19	1.83
Sept. 1985	38370	1940	420	99	18	48
			Inner Sun Bay			
Aug. 1983	740	340	< 1	15	< 5	1.7
Sept. 1983	850	420	< 1	< 10	7	1.6
Aug. 1984	860	490	< 1	< 15	< 5	1.4
Sept. 1984	1160	580	< 1	5	< 5	1.6
Aug. 1985	780	370	< 1	0.6	0.8	0.8
Sept. 1985	18630	1300	190	39	8.5	20.4
			Concession Creek			
Aug. 1983	640	270	< 1	< 10	< 5	1.6
Sept. 1983	720	340	< 1	< 10	< 5	1.9
Aug. 1984	670	330	< 1	< 15	< 5	0.3
Sept. 1984	880	400	< 1	< 5	5	< 0.1
Aug. 1985	670	310	< 1	0.4	0.6	0.8
Sept. 1985	780	380	< 1	0.7	1.0	0.7

Note : 1983 and 1984 data from Reid, Crowther and Partners Limited, 1985.
1985 data from Mudroch and Sutherland, 1988.

Table 1.5: Mean concentrations of metals in sediment samples from Sun Bay collected between 1982 and 1985

Element	Sept. 1982	Sept. 1983	Sept. 1984	Aug. 1985	Sept. 1985
	mg/kg, dry weight				
As	6.2	7.0	7.0	27.8	23.9
Cu	13.7	45.3	23.2	15.6	17.8
Pb	3.1	6.9	4.5	23.3	22.2
Ni	23.9	25.3	20.7	13.4	24.5
Zn	45.9	50.7	39.4	47.6	59.8

1.3.2 Benthos Communities

Benthos data for Contwoyto Lake from 1982 to 1990 were group and analysed. To facilitate comparison between years taxa were grouped into higher taxonomic levels (Families, Orders). Stations 2, 5, 6, and 7 were roughly comparable in terms of water depth and were analysed together. For those stations, data were available for 1985 before and after decant, and for 1990. Although there was no proper control station for contrast with these stations, they were situated along a gradient of exposure to the effluent plume. Based on the the metal concentrations in surface water the predicted gradient of effects between these stations would be 2 ‹ 7 ‹ 5 ‹ 6. A linear orthogonal contrast (Hoke *et al.* 1990) was used to test whether this was true for abundances of each major group of benthos (mites, chironomids, sphaeriids, gastropods, nematodes, oligochaetes, ostracods, trichoptera, miscellaneous diptera), as well as for total abundance. The ANOVA model is given in Table 1.6, while the contrast coefficients are given in Table 1.7. The data indicate that none of the benthic taxa at these sample stations exhibited changes which could be attributable to mining activity

Table 1.6: Analysis of Variance (ANOVA) model testing for relationship between magnitude of change and exposure to effluent for benthic invertebrate taxa

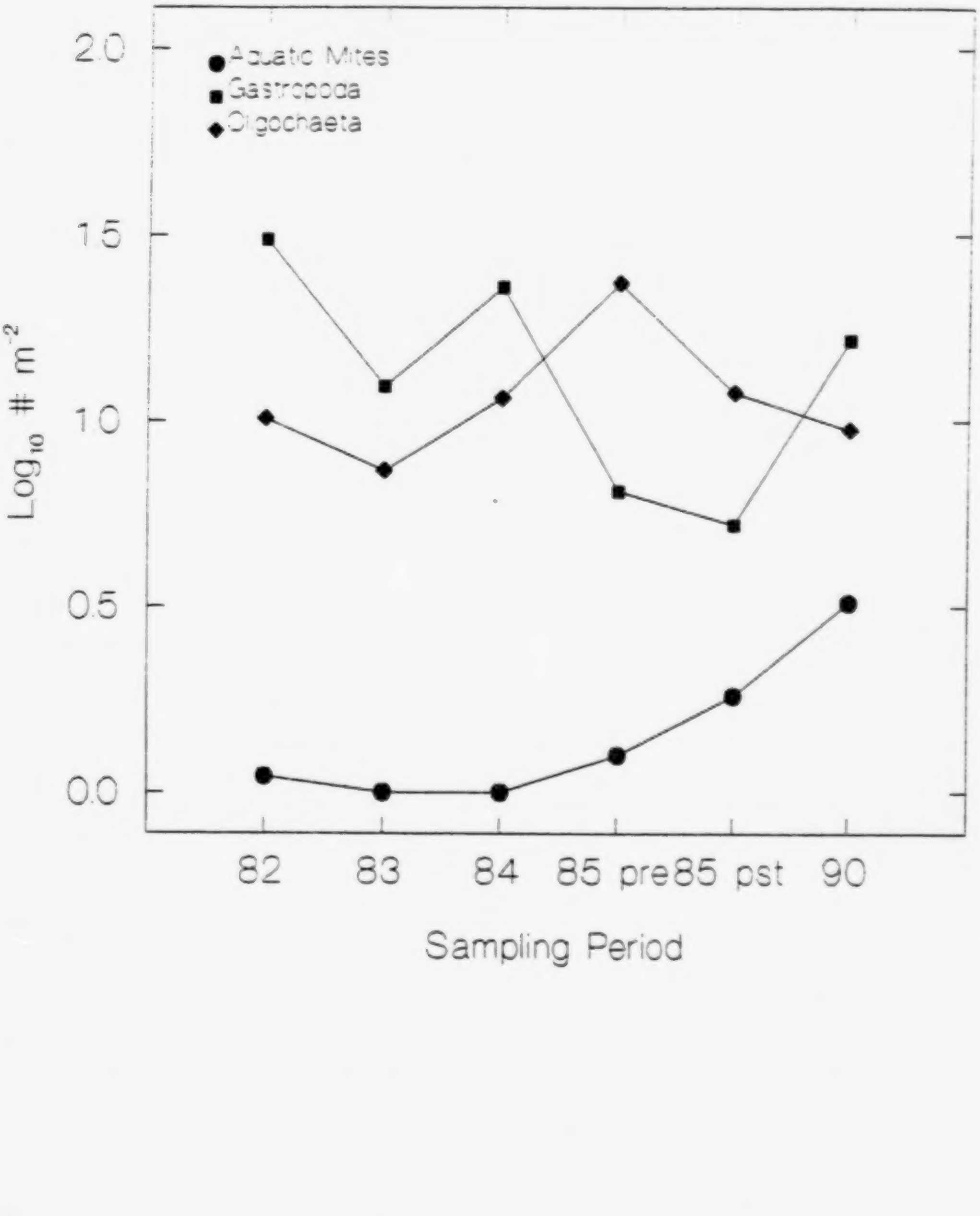
Source	F-test
Stations	contrast - linear effect along the gradient
Time	contrast - before/after decant
Stations - Time	contrast - linear effect - before/after
Residual	MS(contrast)MS(residual)

Table 1.7: Contrast coefficients testing for a linear relationship between magnitude of change and exposure to effluent for benthic invertebrate taxa

Station Number	Time of Sampling	Contrast - before/after decant	Contrast - linear relationship	Contrast - before/after -linear relationship
2	1985 pre-decant	2	-3	-6
	1985 post-decant	-1	-3	3
	1990	-1	-3	3
5	1985 pre-decant	2	1	2
	1985 post-decant	-1	1	-1
	1990	-1	1	-1
6	1985 pre-decant	2	3	6
	1985 post-decant	-1	3	-3
	1990	-1	3	-3
7	1985 pre-decant	2	-1	-2
	1985 post-decant	-1	-1	1
	1990	-1	-1	1

Station 3 was much shallower than the other stations and was treated separately. There was also no control station against which to contrast station 3, consequently, abundances before the first effluent decant were contrasted against abundances after the first decant. Using data from 1982, 1983, 1984, 1985 (before and after decant) and 1990, there were a total of four pre-decant observations and two post-decant observations. Again, a linear orthogonal contrast was used to test for changes in abundances after decant for each of the major taxonomic groups as well as for total abundance. As this station, variations in numbers of aquatic mites and oligochaetes were higher after effluent decant began, and the number of gastropods (snails) were lower (Figure 1.5). These data indicate a change in the benthic community at Inner Sun Bay. Increased numbers of oligochaetes indicate that water quality at that site has been moderately impaired. The increased numbers of mites may also indicate water quality impairment, but little is known about the tolerances of aquatic mites to degraded water quality. The reductions in numbers of snails may indicate a change in the amount of periphytic food available.

Figure 1.5: Abundance of aquatic mites, oligochaetes, and gastropods at Station 3, Inner Sun Bay, in Contwoyto Lake, 1982 to 1990.



1.3.3 Fish Communities

1.3.3.1 Contwoyto Lake Studies

Data for lake trout growth (Ash and Harbicht 1991; R.L. & L. Environmental 1985, 1994) was analysed by analysis of covariance (ANCOVA) of log length versus log age. While significant differences in growth rates were found between years and sites, these were likely due to the relatively large sample size of fish and the narrow size range of the fish sampled. Small, immature fish were absent from the samples, likely as a function of the fishing gear used. No major differences in growth rates were observed, except for fish sampled in 1990 from Contwoyto Lake. The age data for 1990 does not resemble the data for 1982-1984 and 1993, and the 1990 data may be erroneous.

Minor differences also occur in the condition (ANCOVA of log weight versus log length) of lake trout from the three sites. There are no apparent trends for the slopes of regressions of log weight on log length (see Table 1.8) with only the slope from fish sampled from Contwoyto Lake in 1982-84 appearing to be significantly higher. All other slopes are not significantly different than that estimated by Roberge *et al.* (1986) for lake trout caught from Contwoyto Lake in 1981-82, suggesting no changes in condition from baseline studies. Even if differences in growth rate or condition factor due to mining operations existed, the extremely slow growth rates of lake trout in Arctic regions may mask these changes since effects may not be evident in the 10 years since Lupin mine began operations.

Table 1.8: Regression coefficients for log weight versus log length for lake trout captured from Inner Sun Bay, Contwoyto Lake reference area, and Concession Lake

Location	**1982-84**			**1990**			**1993**		
	Regression Coefficient	**Standard Error**	**# of Samples**	**Regression Coefficient**	**Standard Error**	**# of Samples**	**Regression Coefficient**	**Standard Error**	**# of Samples**
Inner Sun Bay	3.164	0.067	41	2.898	0.118	30	2.923	0.134	30
Contwoyto Lake	3.639	0.124	27	3.100	0.086	21	2.922	0.109	20
Concession Lake	2.998	0.071	38	2.912	0.225	15	3.135	0.126	30

In a baseline study, Roberge *et al.* (1986) estimated that male lake trout were completely mature at age 25 years (550-574 mm), while females were completely mature at age 24 years (600-624 mm). It was not possible to estimate the age at maturity from the lake trout data collected during the tissue metals studies (Ash and Harbicht 1991; R.L. & L. Environmental

1989, 1994; Reid Crowther 1985b) due to difficulties in accurately sexing the fish (Rich Pattenden, R.L. & L. Environmental, pers. comm.). The state of maturity of these lake trout was difficult to assess since they may spawn in alternate years, with little gonadal development to help in sexing during the first year after spawning.

Based upon the review and analysis of the available data, it appears that mining operations at the Lupin Mine have not had any significant impact on the growth of lake trout in Contwoyto Lake over the period of operations to date.

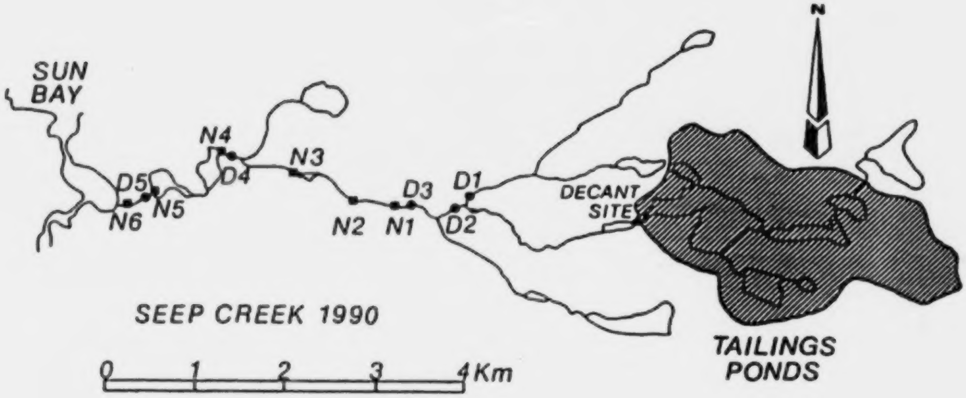
1.3.3.2 Seep Creek Studies

Seep creek has been described as feeding and rearing habitat for Arctic grayling, round whitefish, and lake trout, as well as spawning by Arctic grayling (Reid Crowther 1984). The use of the creek by fish was described in 1983 and 1984 (Reid Crowther 1984, 1985a) and more recently (Hnytka 1991) to evaluate the impact of the tailings discharge on the fish resources of Seep Creek.

Seep Creek originates from a series of small headwater lakes located near the Lupin mine site and flows in a westerly direction, entering Inner Sun Bay. Several of these headwater lakes have now been displaced by tailings ponds (Figure 1.6). The creek flows in a single, well-defined channel over a mixture of boulder, cobble and gravel substrate. The channel widens into two small shallow lakes less than 1 km upstream of the mouth. Peak discharge occurs during the spring freshet, after which flow declines to minimal levels by late summer. Basal flow appears to originate from melting ice. The creek freezes to the bottom in winter along with the shallow lakes in the drainage system. A more detailed description of the creek and characterization of stream flow and bed materials is provided in Reid Crowther (1984).

The data in Table 1.9 indicate a decreased length at age for arctic grayling sampled in Seep Creek in 1990 when compared to pre-operational data from 1983. The 1990 samples were taken about two weeks later in the year, and one possible explanation for the difference in growth my be due to the effects of differing temperature regimes over lifespan of the fish. Due to the different temperature regimes present every year (ice out can vary by up to a month), the length of the growing season may have a much more marked effect on fish size than in southern geographical areas. Another explanation for reduced growth may be the effects of effluent decant and the presence of metals in tea sediments of Seep Creek. However, specific monitoring would be needed to address this possibility. Fewer fish and no mature adults were caught in 1990, but based upon 1983 data, it would appear that the spawning run was completed prior to the commencement of sampling in 1990. The presence of 1 to 4 year-old fish in the 1990 catch would also indicate some successful spawning in the years since the effluent decant began. The absence of 0+ grayling in 1990 is likely a function

Figure 1.6: Map of Seep Creek indicating gill net sites (N 1-6), dishcarge sampling sites (D 1-5) and the approximate location of the tailings, as well as lakes in the area now occupied by tailings.



of the gear used in 1990 (hoop net, gill net, dip net) compared to capture gear used in 1983 (box traps), rather than the complete absence of a year class.

Reduced growth may be due to metals in mine discharges, possible temperature fluctuations resulting from discharges, or due to natural yearly variations in temperature. More data is needed to define natural variability in grayling growth. Additional data is needed to determine if the changes observed in 1990 are due to mining operations or natural variability. Since it is not possible to obtain any additional pre-operational data, it would be necessary to find reference streams flowing into Contwoyto Lake that support grayling populations.

Table 1.9: Age versus fork length (mm) data for Arctic grayling captured in the Seep Creek watershed in 1983 (Reid Crowther 1984) and 1990 (Hnytka)

Year		Age				
		0	1	2	3	4
1983	Mean	45.3	103.5	149.1	189.5	229.9
	N	7	2	10	53	90
	Range	38-50	96-111	139-165	172-210	207-257
1990	Mean	-	80.6	130.6	177.6	223.7
	N	0	13	43	13	6
	Range	-	75-95	110-153	163-190	214-235
	Std. Dev.	-	5.0	9.3	7.0	8.4

1.3.4 Tissue Metal Monitoring Studies

Lake trout from Contwoyto Lake and Concession Lake were sampled over a number of years and the concentrations of arsenic, lead, nickel, copper, cadmium, mercury, and zinc were determined in muscle and liver samples (Ash and Harbicht 1991; R.L. & L. Environmental 1989, 1994; Reid Crowther 1985a). Only mercury residues in muscle and liver increased significantly with body length. Elevated concentrations of mercury in fish tissues may be related to natural geological sources since similar levels of mercury were found in fish collected from reference and exposed sites.

No significant relationships were found to exist between body length and any of the other metals monitored in the studies. In general, for arsenic, nickel, lead, and cadmium, a large proportion of the muscle samples contained concentrations of these metals below detection limits. These metals were detected more often in liver samples, but non-detected values were still present. The lack of relationship with body weight and the high number of values below detection limits for arsenic, lead, nickel, and cadmium suggest that the contamination of fish by these metals is not a problem in Contwoyto Lake. Low concentrations of copper and zinc were found in the muscle and liver of all fish tested, as would be expected of essential minerals.

1.3.5 Toxicity Test Data

Toxicity test data (Table 1.10) demonstrate that effluent was generally not acutely lethal to aquatic organisms and did not elicit a response using the Microtox test.

Table 1.10: Results for toxicity test conducted with rainbow trout, Daphnia magna, and Microtox with effluent from decant stream. Results are reported as % survival in 100% effluent or N.T. (not toxic) for the Microtox test.

Year	Rainbow trout	*Daphnia magna*	Microtox
1994	N.A.	100%	N.T.
1993	100%	100%	N.T.
1992	100%	100%	N.T.
1991	70%	N.A.	N.A.
1990	100%	N.A.	.N.A.
1989	100%	N.A.	N.A.
1988	100%	N.A.	N.A.
1987	100%	N.A.	N.A.
1986	100%	N.A.	N.A.
1985	100%	N.A.	N.A.

N.A. tests were not conducted.
Data from 1990-1994 provided by Anne Wilson, Environment Canada
Data from 1985-1989 provided by Echo Bay Mines Ltd. (1990).

1.4 Discussion and Recommendations

1.4.1 Results of Monitoring

Changes in waterborne metal levels in 1985 were detected as far away from the mouth of Seep Creek as station 2 in Outer Sun Bay. The water quality data presented by Mudroch and Sutherland (1988) demonstrated elevated levels of metals in Seep Creek and Inner Sun Bay after effluent decant began in 1985. Similar increases in metals were not observed in water sampled from Concession Creek at the same time. This would indicate that elevated waterborne metal levels in Seep Creek and Inner Sun Bay were a function of effluent decant and general mining activities. A comparison of waterborne metal concentrations observed in both Inner and Outer Sun Bay and water quality guidelines for the protection of aquatic life is presented in Table 1.11. Copper concentrations exceeded Canadian Water Quality

Guidelines for the protection of aquatic life as far as Outer Sun Bay (CCREM 1987). Zinc and arsenic concentrations exceeded guidelines in samples from Inner Sun Bay, but were below guideline levels in Outer Sun Bay. Nickel concentrations did not exceed guideline levels at any of the stations. However, it is important to bear in mind that data from 1985 cannot be assumed to be representative of more recent conditions, since the 1985 effluent decant was a unique event.

Table 1.11: Comparison of mean metal concentrations in surface water samples collected in September 1985 with the Canadian Water Quality Guidelines

Parameter	Seep Creek	Inner Sun Bay	Outer Sun Bay	CWQGs
		μg/L		
copper	48	20.5	4.7	2.0
nickel	17.7	8.3	2.3	25
zinc	99	38.7	5.4	30
arsenic	421	192.3	41.7	50

Prior to 1985 concentrations of all metals except copper were below Canadian Water Quality Guideline levels. Natural concentrations of copper in this watershed appear to be at the level of the water quality guideline of 2.0 μg/L, which is not unusual for waters flowing through a mineralized area.

Since increased concentrations of metals in water are associated with effluent decant, exposure to metals is periodic in nature, because effluent is only decanted for a short time each summer. This makes it difficult to determine what concentrations of metals organisms are actually exposed to.

The observed changes in numbers of snails, mites and oligochaete worms in Inner Sun Bay after the first decant in 1985 suggested that the effluent from the tailings pond had affected the Inner Sun Bay benthos. The moderate reductions in numbers of snails in Inner Sun Bay was consistent with what would be predicted given that snails are one of the more sensitive groups of aquatic invertebrates (Mackie 1989; Johnson *et al.* 1993). Snails are also sensitive to changes in abundance of periphyton and macrophytes (Mackie and Qadri 1973). Consequently, reductions in numbers of snails may have been caused by sublethal effects of heavy metals or been a secondary response to reduced abundances of plant growth. An increase in the numbers of worms in Inner Sun Bay is also consistent with what would be expected given their tolerances to metals and their ability to colonize habitats where disturbance has occurred (Soster and McCall 1990). Johnson *et al.* (1993) show that aquatic oligochaetes are more tolerant to nickel and zinc than other groups including insects, gastropods and Crustacea. Oligochaetes are, on the other hand, more sensitive to copper than other groups (Johnson *et al.* 1993). It is uncertain whether the increase in numbers of mites is related to the mining effluent. There is little or no literature documenting tolerances of mites to metal mining pollution. However, mites are known to be abundant in organically enriched ponds (Graham *et al.* 1992) and to increase in abundance in acidifying lakes (Brodin and Gransberg 1993).

In general, biological effects of the Lupin Mine on receiving waters are limited to Seep Creek and Inner Sun Bay, while a decreasing metal gradient in sediments was observed as the distance of sample sites from the mouth of Seep Creek increased. The effluent discharge also results in an increased flow and, possibly an increased temperature, in Seep Creek. Interpreting data on Arctic grayling data is difficult since sampling times and fishing gear were not consistent over the years. Future sampling in Seep Creek should ensure that gear is standardize to capture as many year classes as possible and that sampling is conducted over the course of the spawning run so that adults can be captured to estimate reproductive parameters such as fecundity.

The result of the analysis of metals in the livers and muscle of lake trout over the past 12 years is that no consistent, significant relationships or trends have been observed for the metal content of fish muscle and liver, when comparing pre-operational values to tissue metal levels from fish sampled after the mine commenced operations. The only exception is mercury which is present in all samples and exhibits a positive correlation with fish size. Metal residues themselves are an indicator of metal exposure, but if they are not correlated with body weight, for example, they lose any predictive capacity. From a biological monitoring perspective there may be limited value in continuing tissue metal analysis unless a significant link can be established between metal residues and some critical biological endpoint in fish such as growth, mortality, or reproduction. However, in order to ensure that fish are safe to eat, monitoring of metal concentrations in tissue should continue.

1.4.2 Adequacy of Monitoring

1) Statement of hypotheses, development of study design, and selection of reference sites

   As with many environmental monitoring programs that evolve over time, the Lupin Mine monitoring programs could benefit from developing and focusing on specific questions related to aquatic resource protection. Nowhere in any of the documents was a testable hypothesis developed which would focus the studies and specifically test whether or not there were impacts on the aquatic environment receiving mine effluent. If a specific hypothesis or objective is not stated, then the selection of a reference site becomes difficult since it is not clear what parameters are actually being compared. To test for an impact, reference stations should have no exposure to effluent, but be as similar to the exposed locations as possible with respect to "natural" environmental variables. Adequate reference sites were not selected for Contwoyto Lake. In the absence of proper spatial reference sites, pre-operational observations should be collected and compared to observation during the period after the commencement of mining activities (Green 1979). Given that biological systems are variable spatially and temporally, it is necessary to have multiple years of data to properly make an assessment when proper spatial references are not available. The Inner Sun Bay benthos data did provide multiple pre-operational observations that were comparable to the data collected during exposure periods. However, similar data were not available for benthos in the main body of Contwoyto Lake.

2) Selection of sentinel fish species

Seep Creek may provide an excellent site for monitoring the biological effects of the Lupin mine on the receiving environment. Waterborne metal levels were significantly elevated relative to all lake stations monitored and in 1990 there was still a full complement of year classes of arctic grayling. Due to the high conductivity (water hardness) of the water in Seep Creek, metal levels do not appear to be lethal to fish. Although metal levels in the sediments of Seep Creek were not measured, these are likely to be elevated and may have an effect on benthic invertebrate community composition. The use of grayling, rather than lake trout, as a sentinel species has many benefits. Grayling actually spawn in Seep Creek and it is possible to catch all year classes in a given year. The rapid growth of juvenile grayling would allow the monitoring of sublethal impacts in a short time relative to effects on lake trout and would detect problems (if they occurred) before change occurred in Contwoyto Lake. Growth rates of juvenile grayling could also be monitoring without killing the fish due to the distinct length at age relationship. Use of grayling would also eliminate the concern over the possible impacts of destructive sampling of large numbers of lake trout from Contwoyto and Concession Lakes on trout populations in these lakes.

Conversely, since lake trout are an important species to the local aboriginal fishery, at least some sampling of lake trout should continue.

It would appear that the fish tissue metal monitoring program has achieved its goal of demonstrating that, except for mercury, there are no significant, reproducible relationships for copper, arsenic, zinc, or nickel in lake trout and their size or site of capture. Except for the 1990 data set for the Contwoyto Lake site, the growth and condition relationships in lake trout appear to be similar.

1.4.3 Conclusions

Receiving environment effects at the Lupin Mine appear to be restricted to elevated levels of metals in the sediment and water of Seep Creek and Inner Sun Bay of Contwoyto Lake. The benthic invertebrate data suggest a change in the benthic community at Inner Sun Bay, with an increase in numbers of oligochaetes suggesting that water quality at that site has been moderately impaired. Increases in the numbers of mites and moderate reductions in numbers of snails also support the observed changes in oligochaetes. However, it must be remembered that without a proper spatial reference for the Inner Sun Bay station, it is difficult to assess whether the observed changes were natural or due to the mining activity.

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2 Nanisivik Mine

2.1 General Mine Site Description

2.1.1 Site Setting

The Nanisivik lead-zinc mine is located on Strathcona Sound, northern Baffin Island, NWT (73^0 02' N, 84^0 32' W; see Figure 2.1). Strathcona Sound is a deep narrow fjord extending over 50 km from Admiralty Inlet into the Borden Peninsula on Baffin Island (see Figure 2.2). Maximum depths in the sound exceed 300 m in a central basin portion of the sound. This basin is isolated from Admiralty Inlet by a sill at a depth of 150 to 200 m. Strathcona Sound is bounded on both sides by precipitous cliffs often exceeding 300 m in height. The intertidal range in the sound is about 3 m. The largest river which flows into Strathcona Sound is the Strathcona River, which enters on the north side of the inlet. Contributions of water and particulates to the sound by this river are unknown. A number of smaller creeks also flow into Strathcona Sound. One of the longest of these is Twin Lakes Creek (7.2 km), which flows through the mine site.

Topographically, the area is characterized by isolated plateaus, high hills, and broad, deep valleys. The mine rises from sea level at the marine terminal to approximately 650 m above sea level at the airport.

Figure 2.1: Location of the Nanisivik Mine

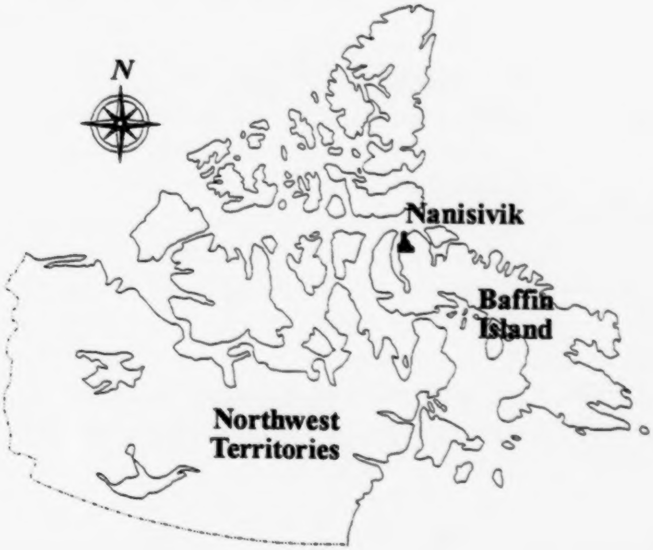
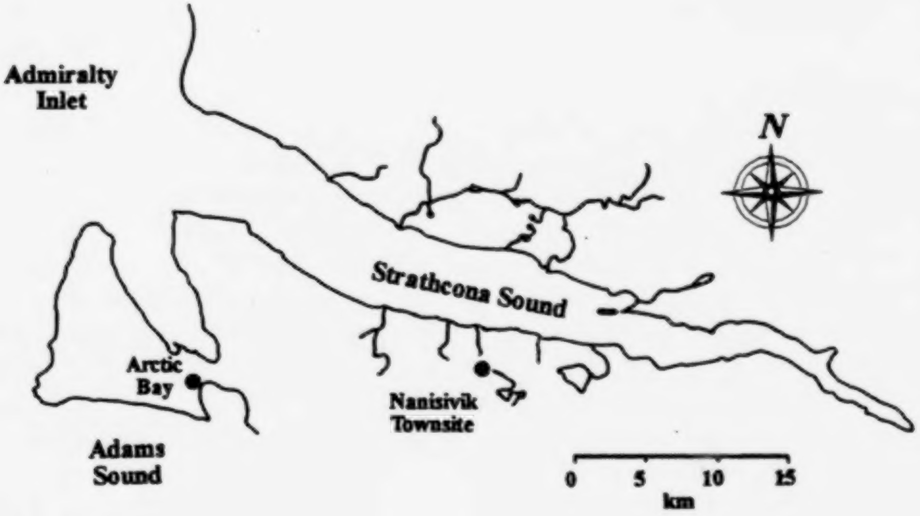


Figure 2.2: Strathcona Sound and Arctic Bay



2.1.2 Geology

The Nanisivik ore body is a rich (about 14% zinc, 1.4% lead) deposit which consists of massive sulphide minerals hosted by the carbonate mineral dolomite ($CaMg(CO_3)_2$). Pyrite (FeS_2) is the dominant sulphide mineral, while the main ore minerals are sphalerite ($(Zn,Fe)S$) and galena (PbS). The main ore body is about 3,000 m long, 100 m wide, and 10 m thick.

2.1.3 History

In June 1974, the Canadian government entered into an agreement with Mineral Resources International where an 18% equity interest in the development of the Nanisivik lead/zinc mine was acquired. The development of this mine was viewed as a "pilot project" to test the feasibility of conducting year-round mining activities in the High Arctic. In order to assess the potential impacts of mine development and operation, the Department of Fisheries and Oceans conducted environmental studies on Strathcona Sound and freshwater lakes in the vicinity of the proposed mine from 1974 to 1976 to gather baseline data on biota and background metal levels. The mine commenced production in October 1976 and in August 1979, the first samples of sediments, algae, invertebrates, and fish were taken from

Strathcona Sound for the determination of trace metals

2.1.4 Mining and Milling Methods

Nanisivik Mines use a combination of open pit and underground mining. The deposit has an open pit at each end, while the central portion is accessed through adits. Ore is crushed underground, reducing it to minus five-inch nominal. The crushed ore is stored underground until it is sent to the mill, located near the western end of the deposit (see Figures 2.3 and 2.4). In the mill, the crushed ore enters a grinding circuit composed of rod and ball mills. The ground ore is then pumped as a slurry to a flotation separation circuit, where methyl isobutyl carbinol, potassium amyl xanthate, and lime are added. The flotation process consists of two circuits, first separating the lead from the ore, then the zinc. The lead and zinc concentrates from the flotation circuit are dried and hauled by truck to the storage shed at the marine terminal, to be shipped out during the open-water season. Waste from the flotation circuit is pumped, as a slurry, to the tailings pond.

2.1.5 Effluent Treatment System

Nanisivik Mines originally proposed that tailings be discharged into Strathcona Sound, at a depth of 46 m, allowing the tailings to settle in a deep area of the sound thought to be devoid of marine life. This proposal was regarded by the proponent as the most economical option, and as the safest method of long-term tailings storage after abandonment of the mine. However, the proposal was not accepted by the Federal Government and an alternate option of discharging tailings into West Twin Lake was proposed. At that time it was estimated that West Twin Lake would provide 20 years of tailings storage capacity.

The current effluent treatment system consists of a tailings pond (West Twin Lake) and a retention pond (see Figure 2.4). The tailings pond is divided into two basins by a dyke. About 375,000 tons of tailings are deposited in basin 1 every year. The tailings are 20 to 50% solids, and contain about 0.1 - 0.2% lead, 0.4 - 0.6% zinc, and 0.01% cadmium, as well as traces of copper, arsenic, iron, and sulphur in both solid and liquid fractions. Initially, tailings were deposited at a depth of three to four metres, but a bathometric survey in 1988 indicated that only about 5 years of containment capacity remained. In the early 1990s the lake reached capacity, and tailings are now deposited subaerially along the west side of the lake.

Particulates settle out in basin 1, and the decant from basin 1 passes to basin 2 where it receives primary treatment to control pH, and lead, zinc, and cadmium concentrations. The decant from basin 2 is received by the retention pond and is further treated with hydrated

Figure 2.3: Nanisivik Mine area

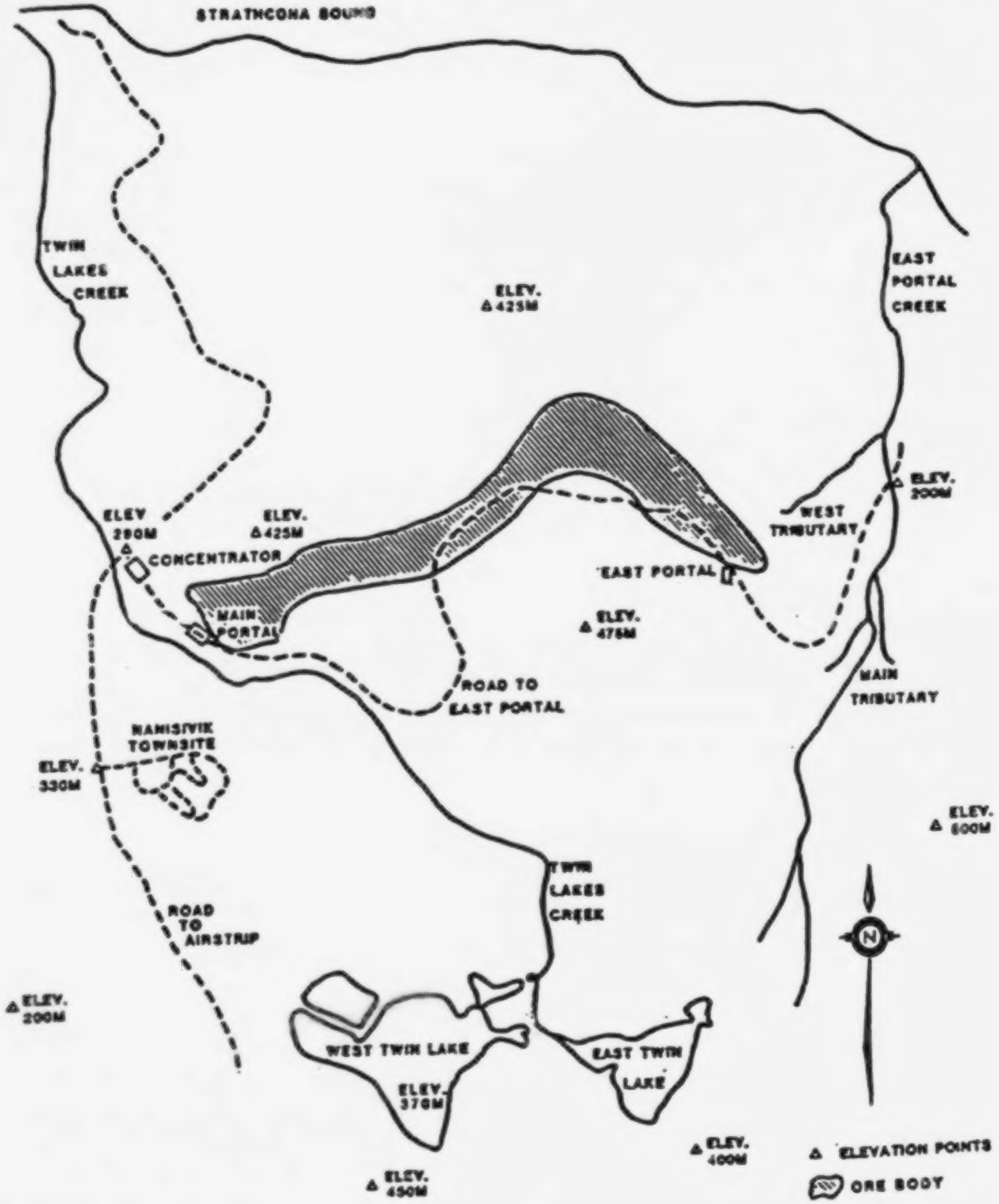
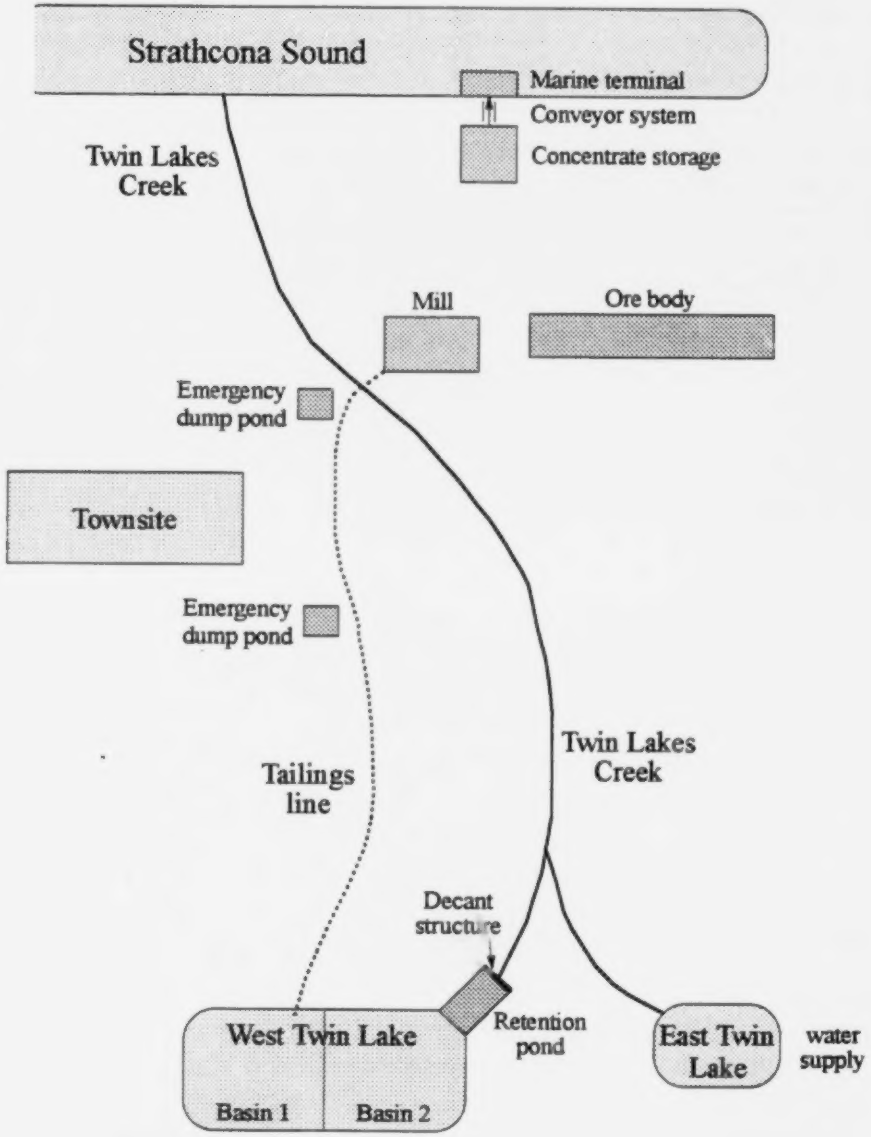

STRATHCONA SOUND
TWIN LAKES CREEK
ELEV. 425M
EAST PORTAL CREEK
ELEV. 200M
WEST TRIBUTARY
ELEV 290M
ELEV. 425M
CONCENTRATOR
EAST PORTAL
MAIN PORTAL
ELEV. 475M
ROAD TO EAST PORTAL
MAIN TRIBUTARY
NANISIVIK TOWNSITE
ELEV. 330M
ELEV. 600M
TWIN LAKES CREEK
ROAD TO AIRSTRIP
N
ELEV. 200M
WEST TWIN LAKE
ELEV. 370M
EAST TWIN LAKE
ELEV. 400M
ELEV. 450M
ELEVATION POINTS
ORE BODY

Figure 2.4: Schematic map of Nanisivik Mine area


Strathcona Sound
Marine terminal
Conveyor system
Concentrate storage
Twin Lakes Creek
Mill
Ore body
Emergency dump pond
Townsite
Emergency dump pond
Twin Lakes Creek
Tailings line
Decant structure
Retention pond
West Twin Lake
Basin 1
Basin 2
East Twin Lake
water supply

lime prior to discharge to Twin Lakes Creek. Decant flow is controlled so that it can be stopped if water quality is approaching NWT Water Board licence limits.

The eastern portion of the ore deposit is drained by East Portal Creek. Runoff from the open pit at the east end of the deposit, and runoff from areas of exposed sulphide rich outcrop is collected and treated with lime in the East Adit Retention Pond prior to discharge.

2.1.6 Water Licence Requirements

The quality and quantity of wastewater discharges from the Nanisivik Mine are regulated by a Water Licence (N5L3-0159) issued under the *Northern Inland Waters Act* and administered by the Department of Indian Affairs and Northern Development. The Water Licence has undergone two amendments and three renewals since the licence was originally granted on July 5, 1976. The amendments and renewals can be summarized as follows:

1) Amendment to increase water consumption was applied for on May 31, 1977, and granted on October 12, 1977, to be effective on June 1, 1977.

2) An application was made on December 15, 1977 for the renewal of the licence which was to expire May 1, 1978. The renewal was for five years and required lower Maximum Average Concentrations (MAC) of lead (reduced from 0.2 mg/L to 0.1 mg/L) and zinc (reduced from 0.4 mg/L to 0.2 mg/L),[2] and allowed for increased water use. The licence also required that a study on the stabilization of the West Twin Lakes tailings area be conducted.

3) An application was made on December 3, 1982 for the renewal of the licence which was to expire April 30, 1983. The renewal was for five years, with the company requesting an increase in the licence limit for cadmium and the introduction of mercury sampling, with both requests being denied.

4) Amendment to increase allowable potable water consumption from 10,000 m^3/month was applied for on January 28, 1986. The application for amendment was changed by the Water Board into renewal of the licence, and after a public hearing, was granted on January 13, 1987.

5) An application was made for the renewal of the licence which was to expire October 31,

---

[2] Maximum monthly arithmetic mean concentrations of lead and zinc allowable under the MMLER are 0.2 mg/L and 0.5 mg/L, respectively.

1991. The renewal was for five years, with an expiry date of October 31, 1996.

The current Water Licence specifies that all waste discharged from the Retention Pond into Twin Lakes Creek shall meet the following effluent quality standards presented in Table 2.1. Non-compliance events are summarized in Table 2.2.

Table 2.1: Effluent quality standards for discharge water from retention pond

Parameter	Maximum Average	Max. Concentration of any Grab Sample (mg/L)
total arsenic	0.25	0.5
total cadmium	0.005	0.01
total copper	0.1	0.2
total lead	0.1	0.2
total zinc	0.25	0.5
oil and grease	15	30
suspended solids	15	30
pH (minimum)	> 6.0	> 6.0

Table 2.2: Summary of non-compliance events

Date	Event
1977	MAC for cadmium was exceeded
1978	MAC for lead was exceeded on one occasion
1980	MAC for lead was exceeded on one occasion
1984	MAC for lead was exceeded on one occasion
1985	MAC for lead exceeded on 19 occasions. The maximum grab concentration for lead was exceeded on three occasions
Aug. 12-14, 1985	MAC for cadmium was exceeded
1986	MAC for lead was exceeded
1987	MAC for lead was exceeded
1987	MAC and maximum grab concentrations for cadmium were exceeded on three occasions
1988	MAC for lead was exceeded on 7 occasions
Aug. 19 to Sept. 15, 1988	six instances of non-compliance

2.1.7 Other Potential Sources of Contamination

Acid Mine Drainage

Downstream from the retention pond, Twin Lakes Creek passes several areas of mineralised outcrops. As a result of the oxidation of sulphide minerals, seeps occur in these areas which have low pH (1.7 - 2.2), high conductivity (25,000 - 29,000 μmhos), and high concentrations of metals and sulphate, but are generally of low volume (Grey 1989).

Dust

There are concerns about dust blowing off the tailings, particularly in the winter when the tailings surface is freeze-dried and more susceptible to wind erosion. Some dust may also be contributed by trucks which haul concentrate to the concentrate storage facility, and by activities at the ore loading facility during the shipping season. Snow in the vicinity of the tailings is frequently blackened with dust, and dust occasionally falls in the community of Arctic Bay. As well as being a potential contaminant for terrestrial ecosystems, the dust may contaminate surface waters, including Twin Lakes Creek and Strathcona Sound.

2.2 Quantity and Quality of Data Available for the Site

The data available for Strathcona Sound in the vicinity of Nanisivik Mine is limited to water and sediment quality data and metal residues in marine plants, invertebrates, fish, and mammals. Limited data was available for fish population structure (sculpins) and apparently only one zooplankton and benthic invertebrate survey was conducted.

2.2.1 Assessment of Data Quality

2.2.1.1 Water Quality

Water quality data is available for both freshwater and marine environments in the vicinity of the Nanisivik Mine, throughout the operational history of the site. Although most of the studies provided detailed methodology for collection and analysis of samples, some of the studies were lacking in this respect (Nanisivik Mines Ltd. 1994). Samples appear to have been taken with the necessary precautions (eg. polyethylene or Teflon sampling apparatus), analysed with standard techniques (flame or graphite furnace atomic absorption spectrophotometry), and accuracy and precision assessed using standardized reference material.

2.2.1.2 Sediment Quality

Surficial sediment sample surveys were conducted in Strathcona Sound on six occasions prior to the commencement of mining activities in 1976. Following the commencement of mining operations, sediment samples were collected in August 1979 (Fallis 1982) and in May/June 1982 (Thomas and Erickson 1983). Although metal levels in sediments were determined using acceptable techniques and QA/QC, comparison of data between studies is difficult due to the different size fractions of sediment that were analysed for metals in each case. Data on sediment quality in Twin Lakes Creek are not available.

2.2.1.3 Benthic Invertebrates

The only data provided on the structure of the benthic invertebrate community was by B.C. Research (1975). These baseline studies also assessed the zooplankton community in Strathcona Sound. Rather than assess benthic invertebrate community structure, all other studies have focussed on analysing metals levels in specific invertebrates such as the clam (*Mya truncata*) or the purple sea urchin (*Strongylocentrotus droebachiensis*).

2.2.1.4 Fisheries

No fisheries community assessment has been conducted in Strathcona Sound. Fisheries studies have been restricted to the collection of sculpins (*Myoxocephalus* spp.) and arctic char (*Salvelinus alpinus*) and subsequent analysis of muscle and liver for heavy metal content. Length and weight data have been collected coincidentally to determine relationships between tissue metal content and weight. No data on the age or population structure of fish from Strathcona Sound are available.

2.2.1.5 Seals

Wagemann (1985) determined arsenic, selenium, and heavy metal levels in the liver, kidney, and muscle of ringed seals captured from the Strathcona Sound area, or from a reference area (Admiralty Inlet) in 1984.

2.2.1.6 Plants

In addition to animals, various studies examined the metal content of seaweed species found in Strathcona Sound. Data are not available on community structure or distribution, as samples were collected solely for the purposes of measuring heavy metal content.

2.2.2 Adequacy of Monitoring Programs

None of the studies clearly stated an objective or testable hypothesis in the design of their monitoring programs. The choice of sample locations was logical, examining the possibility of a gradient in response parameters both east and west in the sound, as well as on the north side of the sound. Some studies included a reference site in Arctic Bay, a distinct bay south of the Twin Lakes Creek watershed (see Figure 2.2).

The Nanisivik monitoring program is based primarily on measuring the distribution and concentration of cadmium, lead, and zinc in the water of Twin Lakes Creek and the water and sediments of Strathcona Sound. The examination of effects of metal contamination on the biota of the sound is limited to determining the metal content of various tissues in fish, benthic invertebrates, and plants. Only one study (Thomas *et al.* 1983) attempted to relate growth and physiology of the *Mya truncata* to tissue metal levels. Although no meaningful relationships could be established between metal levels and physiological function, the intent was sound. The size and volume of Strathcona Sound and the relatively small extent of metal-loading would make using a sentinel fish species quite difficult since only a few species are present and the fish are too mobile and may move amongst the gradients of metal-loading in the sound. To date no examination of the benthic invertebrate community structure has been conducted. Benthic invertebrates tend to be relatively sessile and due to the marked decrease in the metal content of sediments with distance from the mouth of Twin Lakes Creek, they may provide the most effective method for assessing actual biological effects of metals in the sound. It may also be possible to assess benthic invertebrate community structure in Twin Lakes Creek and compare that with other creeks in the vicinity to determine differences in effects between freshwater and marine systems.

2.3 Results of Environmental Monitoring Studies

2.3.1 Water Quality

2.3.1.1 Twin Lakes Creek

The main contaminant of concern in the surface water of Twin Lakes Creek is zinc. Ranges in zinc concentrations from water samples collected at the mouth of Twin Lakes Creek are summarized in Table 2.3. During the 1991 to 1994 sampling period annual total loading of zinc in Twin Lakes at the mouth of the creek ranged from 1700 to 3280 kg (Nanisivik Mines Ltd. 1994).

Table 2.3: Summary of zinc concentrations in water samples collected from the mouth of Twin Lakes Creek

Time Period	Range of Zinc Concentrations	Reference
1974	223 ± 111μg/L (n=3)	Fallis 1982
1975	236 ± 113 μg/L (n=5)	Fallis 1982
1976	47 ± 8 μg/L (n=3) 63 ± 23 μg/L (n=4)	Fallis 1982
1977	341 ± 149 μg/L (n=15)	from Nanisivik Mines Ltd. Annual Reports
1978	147 ± 49 μg/L (n=3)	from Nanisivik Mines Ltd. Annual Reports
1979	583 ± 184 μg/L (n=8)	from Nanisivik Mines Ltd. Annual Reports
1980	2500 ± 1210 μg/L (n=11)	from Nanisivik Mines Ltd. Annual Reports
1991 to 1994	76 to 4190 μg/L	Nanisivik Mines Ltd. 1994

A recent study (1994) conducted as a requirement of the Water Licence examined the annual discharge of metals from Twin Lakes Creek into Strathcona Sound from 1991 to 1994 (Nanisivik Mines Ltd. 1994). Weekly samples taken from 14 sample stations in the watershed during the open water season (late May to mid September) and were analysed for total zinc, lead, cadmium, total suspended solids, pH, conductivity, and temperature. Zinc concentrations in the creek increased dramatically downstream of the West Pit, particularly during periods of high precipitation.

Lead, cadmium, arsenic, mercury, iron and copper concentrations have also been monitored in Twin Lakes Creek. At the mouth of the creek, from 1976 to 1980, all were consistently significantly less than the maximum average concentrations in tailings pond decant, stipulated in the water licence for the operation. Cadmium and lead concentrations were consistently low at all sample stations in the creek during the 1991 to 1994 monitoring program.

2.3.1.2 East Portal Creek

Water samples from a station in East Portal Creek (see Figure 2.2) contained elevated levels of zinc, but decreased consistently from 1984 to 1988 after the implementation of a treatment pond on the creek (Grey 1989). High zinc levels in this creek were also due to acid production and leaching through mining-exposed surface areas and waste rock deposited in and beside East Portal Creek.

2.3.1.3 Strathcona Sound

Pre-operational data for metal concentrations in surface waters in Strathcona Sound are reported in B.C. Research (1975) and Fallis (1982) and presented in Table 2.4.

Table 2.4: Pre-operational metal concentrations in surface water samples from Strathcona Sound

Metal	B.C. Research 1975* (μmol/m^3)	Fallis 1982 (μg/L)
lead	214 - 640	2.0
zinc	< 24	6.2
cadmium	< 9	< 0.2
copper	n.a.	1.0

* sampling, storage, and handling techniques for these water samples may have resulted in contamination of the samples, so these data should be viewed with some caution (Seakem 1989).
n.a. - not analysed

Thomas *et al.* (1983) collected samples from five locations in the sound for metal analysis. Zinc and lead concentrations were 4-times higher in samples collected 0.2 km from the mine than in samples collected further from the mine, although there were no trends in concentrations of samples collected further from the mine.

A study of water chemistry in the sound in the winter and spring of 1984 and 1985 reported the following (after Arctic Laboratories Ltd. 1986):

- **Cadmium:** concentrations showed little variation with depth or location in the sound, and were at or below background concentrations. Cadmium concentrations ranged from 0.03 to 0.1 μg/L.
- **Zinc:** concentrations in December 1984 showed no significant gradient moving away from the site (concentrations ranged from 0.33 to 0.63 μg/L in samples from the station closest to the marine terminal, and from 0.16 to 0.38 μg/L at a station further offshore). However, in June 1985 there was a significant gradient (concentrations ranged from 3.1 μg/L at the station closest to the terminal to 0.16 μg/L at a station further offshore).
- **Lead:** concentrations in the sound in June 1985 were an order of magnitude higher than those typically found in open oceans. Lead concentrations in surface waters in the sound were two to five times higher at the station closest to the marine terminal than at a station further offshore. Lead concentrations decreased with depth in the water column.
- **Iron:** no significant gradient between sites was reported in the 1984 and 1985 sampling. Iron concentrations at all stations were similar to or less than concentrations typically found in open oceans.

Moore (1983) reported different and in some cases conflicting results. Iron concentrations in the sound ranged over almost two orders of magnitude with the lowest concentrations many times higher than open-ocean levels. Cadmium concentrations ranged from concentrations similar to mean oceanic concentrations, to as much as 400-fold higher than concentrations measured in the Central Arctic Ocean (Moore 1983). Concentrations of lead, zinc, and iron were elevated approximately 100-, 25, and 2-fold, respectively, in surface water in the vicinity of the mouth of Twin Lakes Creek and the marine terminal/ore concentrate loading dock compared to surface water concentrations at mid-inlet stations.

2.3.2 Sediment Quality

Surficial sediment levels of lead, zinc, and cadmium are summarized by Seakem (1989). Pre-operational levels of lead and zinc at some locations in Strathcona Sound are higher than those considered typical of uncontaminated marine sediments (Table 2.5). Cadmium concentrations are essentially uniform over the entire sound, while the distribution of lead and zinc were more variable, with higher concentrations occurring on the south side near the mouth of Twin Lakes Creek (B.C. Research 1975).

Sediment samples taken from Strathcona Sound in 1979 (Fallis 1982), 1980-81 (Fallis 1990), and 1982 (Thomas and Erickson 1983) provide data on metal levels after the mine commenced operation (sampling stations of Fallis are shown in Figure 2.5). Levels of lead, zinc and cadmium in the surficial sediments of Strathcona Sound appeared to be higher compared to pre-operational sampling or the Arctic Bay reference station (Fallis 1982), with the highest levels occurring near the mouth of Twin Lakes Creek (station 3). However, metal levels higher than background were reported during pre-operational sampling from this area, suggesting that sediments in Strathcona Sound have a natural source rich in these metals. Levels of all three metals were also found to increase to the interior of the fjord, either due to current flow or contributions of metals from natural geological conditions (Fallis 1990). Lead, zinc, and cadmium levels at station 3 (marine terminal area) were elevated in 1979 relative to pre-operational conditions (Fallis 1982), but decreased in 1980-81 samples (Fallis 1990). Lead and zinc concentrations in sediment samples collected from 1979 to 1981 are plotted against distance from the mine in Figure 2.6.

Figure 2.5: Location of sampling stations in Fallis (1982)

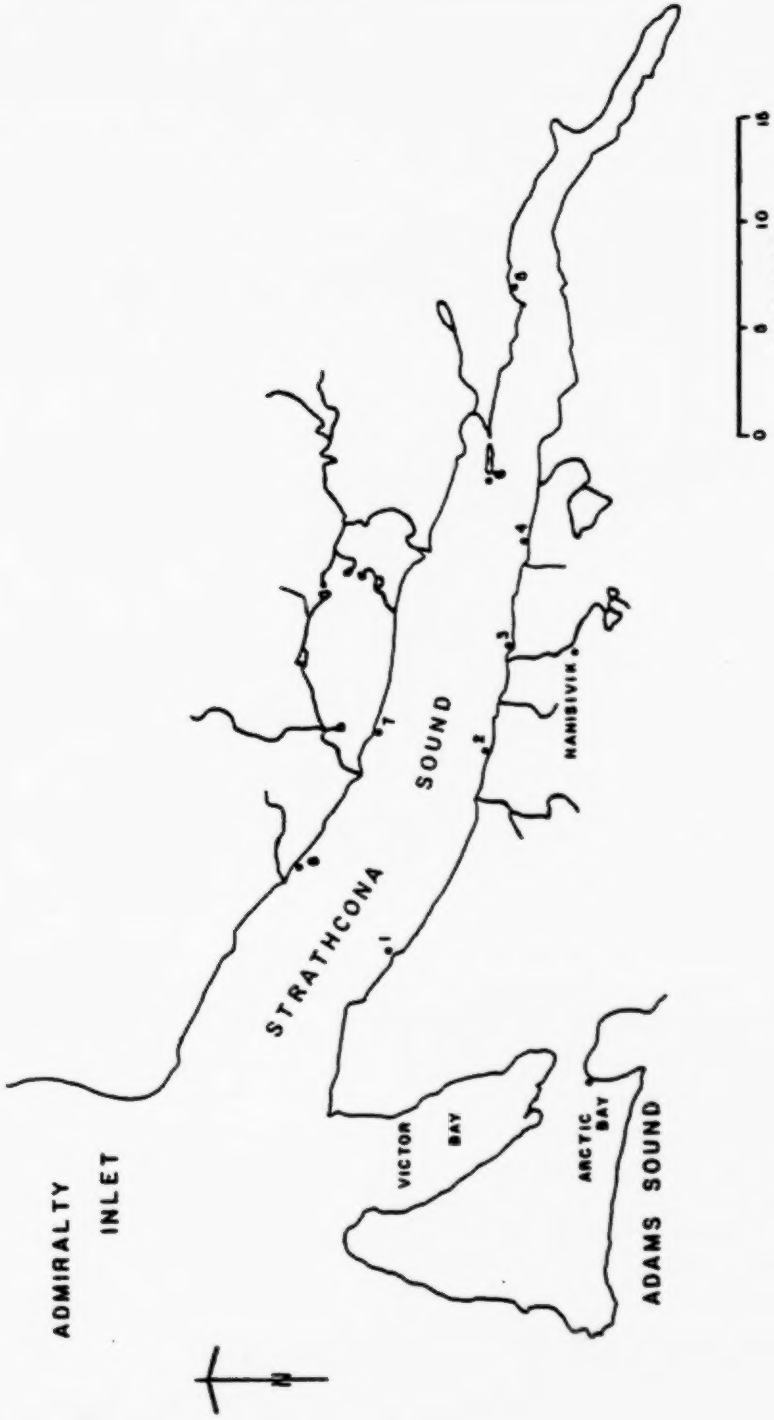
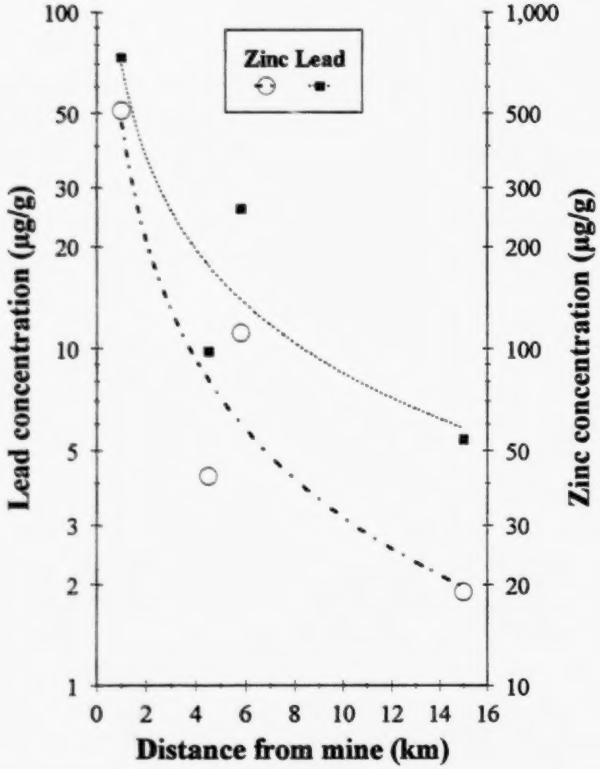

STRATHCONA SOUND
NANISIVIK
VICTOR BAY
ARCTIC BAY
ADMIRALTY INLET
ADAMS SOUND
1
2
3
4
7
0
5
10
15
KILOMETRES
N

Table 2.5: Summary of lead, zinc, and cadmium content of surficial sediment samples collected in Strathcona Sound and Arctic Bay from 1974 to 1981

Location	Station Number (1)	Date Collected	Sediment fraction analysed	Pb μg/g	Zn μg/g	Cd μg/g
Arctic Bay		Aug. 76 (2)	< 180 μm	< 0.25	61.3	< 0.25
Strathcona Sound	3	Aug.75 (2)	< 180 μm	18	175	0.2
		Aug. 79 (2)	< 1 mm	76	750	1.8
		1980 (3)	?	64.5	398	1.35
		1981 (3)	?	79.1	379	2.15
	2	Sept. 75 (2)	< 2 μm	20	125	< 0.2
		Sept. 75 (2)	whole	10	50	< 0.2
		Aug. 79 (2)	< 1 mm	5.6	23	0.03
	4	Sept. 75 (2)	< 2 μm	20	110	< 0.2
		Sept. 75 (2)	whole	20	66	< 0.2
		Aug. 79 (2)	< 1 mm	9.6	94	0.18
	7	Sept. 75 (2)	whole	10	53	< 0.2
		Aug. 79 (2)	< 1 mm	10	38	0.02
	8	Sept. 75 (2)	< 2 μm	20	110	< 0.2
		Sept. 75 (2)	whole	15	65	< 0.2
		Aug. 79 (2)	< 1 mm	10	34	0.02
	1	Sept. 75 (2)	whole	10	72	< 0.2
		Aug. 79 (2)	< 1 mm	3.0	20	0.05
	5	Sept. 75 (2)	< 2 μm	46	100	< 0.2
		Sept. 75 (2)	whole	< 1	65	< 0.2
		Aug. 79 (2)	< 1 mm	6.5	40	0.03
	?	Aug. 75 (4)	< 180 μm	2-23	40-200	0.1-0.2
			< 2 μm	20-46	100-125	< 0.2
			whole	1-20	50-72	< 0.2
	?	Sept. 75 (4)	< 180 μm	< 1-30	3-85	< 0.2
			< 2 μm	15-30	80-135	< 0.2
typical uncontaminated (4)			whole	0.1-15	5-144	0.02-0.6

(1) Station locations are shown in Figure 5
(2) Fallis 1982
(3) Fallis 1990. These figures are means. Sample sizes and standard deviations are not known
(4) after Seakem 1989

Figure 2.6: Lead and zinc concentrations in sediment samples collected from 1979 to 1981, versus distance from the Nanisivik Mine



Thomas *et al*. (1983) also examined sediment chemistry, and reported that concentrations of lead, zinc, and cadmium in the sediments were highest at the surface and decreased with depth along sediment cores. The metal concentrations in the sediments are generally higher than those usually reported for uncontaminated coastal marine sediments. The spatial and vertical distribution of metals in the sediments of Strathcona Sound suggests a source of metals adjacent to the mouth of Twin Lakes Creek (Thomas *et al.* 1983).

2.3.3 Biological Monitoring in Strathcona Sound

Several studies examining the metal content (lead, zinc, cadmium) of fish and invertebrates in Strathcona Sound were conducted prior to the commencement of mining operations in

September 1976 (B.C. Research 1975; Bohn and Fallis 1978; Bohn 1979; Fallis 1982; Thomas *et al.* 1983).

2.3.3.1 Fish

Bohn and Fallis (1978) summarized normal background metal concentrations in shorthorn sculpins and Arctic char in Strathcona Sound in 1974. Muscle and liver tissues were analysed for arsenic, cadmium, copper, lead and zinc. All fish were weighed and measured (fork length). A significant positive correlation was observed between arsenic concentration in sculpin muscle and liver and body weight. Data on fish after the mine began operating are limited to a few samples collected in 1979, from which no conclusions can be drawn.

2.3.3.2 Invertebrates

Clams (*Mya truncata*)

Fallis (1982) and Thomas *et al.* (1983) reported data on the clam *Mya truncata.* Their work was briefly summarized in Asmund (1990). Fallis (1982) and Thomas *et al.* (1983) measured the length and width of the clams, and the soft tissues were analysed for metals. In addition, Thomas *et al.* measured condition indices (ratio of dry weight of clam tissue / volume of enclosed shell), morphometrics (length, width, breadth of left valve), and proximate composition (moisture content, protein, lipid). Glycogen was determined as a measure of carbohydrate energy storage and free sugars were measured as an indication of impairment of metabolism. Thomas *et al.* (1983) also used histological techniques to examine and compare the reproductive capacity and physical conditions of clams from Strathcona Sound and Arctic Bay. Reproductive capacity was assessed by comparing maturity of gametes and the condition of follicles.

Lead and zinc concentrations in the tissue of *Mya truncata* are summarized in Table 2.6 and concentrations are plotted against distance from the mine in Figure 2.7.

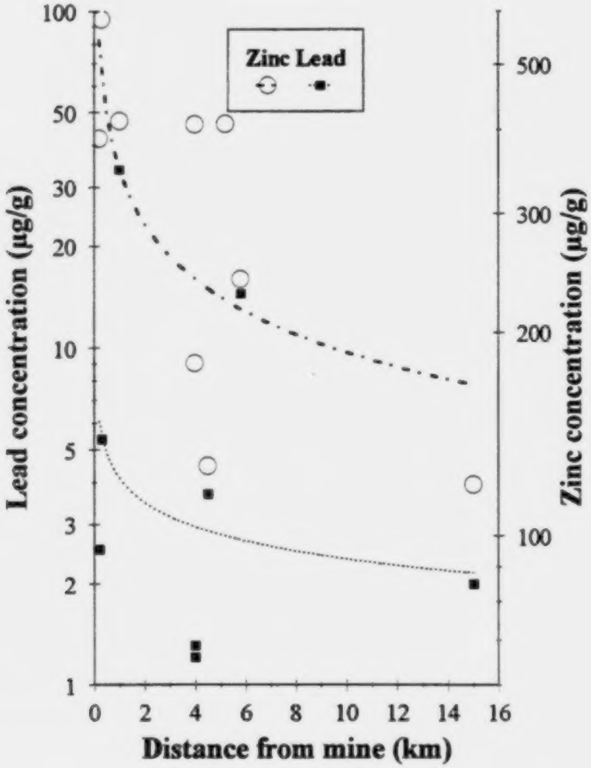
As is evident from Figure 2.7 zinc concentrations are significantly higher in samples collected closest to the mine. Data are more variable for lead, although Fallis (1990) concluded that lead concentrations in clams were higher at station 3 (closest to the mine) than at other stations further from the mine (62.8 to 67.6 μg/g at station 3, versus a range of 2.42 to 22.9 μg/g at all other sites). Fallis (1990) also reported that cadmium concentrations in tissue of *Mya truncata* were highest towards the mouth of the sound.

Table 2.6: Summary of lead and zinc concentrations in tissue of Mya truncata collected from Strathcona Sound (after Asmund *et al.* 1990)

Station number	Distance from mine (km)	Zinc	Lead	Reference
		μg/g		
3	1	437	65.4	Fallis (1990)
		388	2.28	Fallis (1982)
2	4.5	137	6.3	Fallis (1990)
		118	1.00	Fallis (1982)
4	5.8	193	22.1	Fallis (1990)
		288	6.96	Fallis (1982)
1	15	114	2.7	Fallis (1990)
		124	1.42	Fallis (1982)
DS5	0.2	388	2.53	Thomas *et al.* (1983)
DS3	0.3	584	5.38	Thomas *et al.* (1983)
DS2	4.0	407	1.24	Thomas *et al.* (1983)
DS4	4.0	180	1.31	Thomas *et al.* (1983)
DS1	5.2	408	0.72	Thomas *et al.* (1983)
DS6	Arctic Bay	273	2.46	Thomas *et al.* (1983)

Thomas *et al.* (1983) found that no relationship between metal burden and body weights could be determined due to the limited sample size. In addition, no apparent relationship was observed between the health of clams (condition index, biochemistry, histology) and their proximity to the mouth of Twin Lakes Creek. Thomas *et al.* also reported that condition, proximate composition and growth were significantly different for samples collected from Strathcona Sound than for samples collected from the reference site in Arctic Bay. Dry weight, condition index, lipid and glycogen content were lower in Arctic Bay specimens relative to some Strathcona Sound animals, likely due to post-spawn sampling in Arctic Bay (samples were collected at the same time, but the populations were in different reproductive phases). No differences in protein levels were found between sites. Histology indicated no morphological evidence of reproductive impairment in clams from Strathcona Sound; no obvious or well-developed pathologies were evident in any tissues.

Figure 2.7: Lead and zinc concentrations in the tissue of *Mya truncata*, versus distance from the Nanisivik Mine



Sea Urchin (*Strongylocentrotus droebachiensis*)

Bohn (1979) examined metal levels in the sea urchin *Strongylocentrotus droebachiensis* from six sites varying in proximity to the mouth of Twin Lakes Creek. Iron and zinc levels were higher in whole sea urchins within 1 km (n = 26) of the stream mouth compared with a site 15 km away, while other metals showed no trends. Concentrations ranges in whole sea urchins are presented in Table 2.7.

Fallis (1982) reported that lead and zinc concentrations in sea urchins collected at station 3 (closest to the mine) were significantly higher in samples collected in 1979 than in samples from 1976 (5-times higher for zinc). In 1979 lead and zinc concentrations in samples from station 3 were significantly higher in samples from stations further from the mine.

Table 2.7: Metal concentrations in tissue of sea urchin *Strongylocentrotus droebachiensis* collected at different distances from the mouth of Twin Lakes Creek (after Bohn 1979)

Distance from Twin Lakes Creek (km)	As	Cd	Cu	Fe	Zn
	ppm				
1	2.7 - 4.5	0.9 - 1.8	2.3 - 4.0	570 - 2610	22 - 78
15	3.4 - 4.2	1.0 - 1.4	2.4 - 3.3	305 - 845	29 - 39

3.3.3.3 Macrophytes

Bohn (1979) examined metal levels in seaweed *Fucus distichus* from six sites varying in proximity to the mouth of Twin Lakes Creek. Only zinc was significantly higher in whole *Fucus distichus* from within 0.1 km of the mouth of Twin Lakes Creek compared to other sites at least 1 km away. Older tissues contained the highest concentrations of zinc, copper, and iron, while new, actively growing tissue contained the highest concentrations of cadmium. Bohn's (1979) data are summarized in Table 2.8.

Table 2.8: Metal concentrations in the tissue of *Fucus distichus* collected at different distances from the mouth of Twin Lakes Creek (after Bohn 1979).

Distance from Twin Lakes Creek (km)	As	Cd	Cu	Fe	Pb	Zn
	ppm					
0.1	22 - 29	0.9 - 1.8	2.5 - 4.7	285 - 395	n.a.	106 - 170
1.0	26 - 30	1.0 - 1.4	2.2 - 4.5	505 - 920	0.5 - 1.4	50 - 110

Fallis (1982) conducted studies of metal concentrations in seaweed in 1976 and 1979. Samples were also collected from Arctic Bay (1976) to serve as controls. During the 1979 study three species of seaweed (*Fucus vesiculosus, Laminaria solidungula, Agarum cribrosum*) were collected from 8 stations. Metals were measured in new and old growth of seaweed. Data for *Fucus vesiculosus* are summarized in Table 2.9. Mean lead and zinc concentrations versus distance from the mine for *Fucus vesiculosus* collected from 1979 to 1981 are plotted in Figure 2.8.

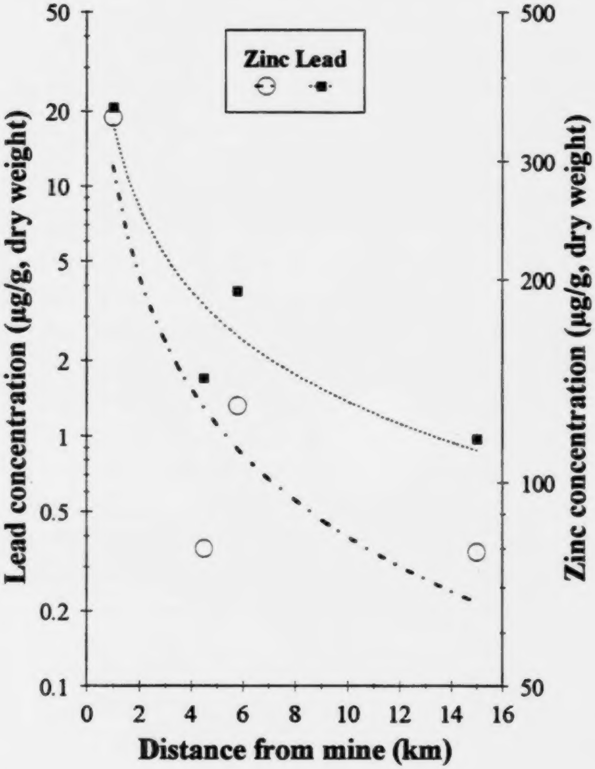
Table 2.9: Summary of lead, zinc and cadmium concentrations in the tissue of *Fucus vesiculosus* (total plant, washed) collected in 1976 and 1979 (Fallis 1982).

Location	Station Number (1)	Date Collected	# of Samples	Pb	Zn	Cd
					μg/g	
Arctic Bay		Aug. 76	8	2.11 ± 0.83	23.8 ± 9.68	1.70 ± 0.61
Strathcona Sound	3	Aug.75	2	2.10 ± 0.06	88.4 ± 3.32	0.63 ± 0.06
		Aug. 79	1	59.5	493	1.66
		1980 (1)	?	?	379	1.21
		1981 (1)	?	?	458	1.61
Strathcona Sound	2	Aug. 79	5	1.88 ± 0.18	67.0 ± 4.71	1.82 ± 0.88
Strathcona Sound	4	Aug. 79	4	6.66 ± 1.24	141 ± 10.6	0.88 ± 0.05
Strathcona Sound	7	Aug. 79	1	2.45	47.8	0.67
Strathcona Sound	6	Aug. 79	1	1.64	93.7	1.28
Strathcona Sound	8	Aug. 79	1	1.41	36.5	1.12
Strathcona Sound	1	Aug. 79	2	1.19 ± 0.08	52.2 ± 3.32	1.66 ± 0.14
Strathcona Sound	5	Aug. 79	2	1.08 ± 0.13	58.1 ± 0.17	0.94 ± 0.04

(1) 1980 and 1981 data are for old growth tissue only, rather than total plant

In 1976 zinc levels were 4-fold higher in the seaweed *Fucus vesiculosus* from Strathcona Sound compared to plants from Arctic Bay. Only one sample was collected at station 3 in 1979, but the lead, zinc and cadmium concentrations in this sample were 28, 21 and 2.5 fold higher than the mean concentrations in samples from the same location in 1976. Metal concentrations in the sample from station 3 were also substantially higher than concentrations in samples collected from stations further from the mine site.

Figure 2.8: Lead and zinc concentrations in the tissue of *Fucus vesiculosus*, versus distance from the Nanisivik Mine



2.3.3.4 Seals

Examining the heavy metal levels in kidney, liver, and muscle of ringed seals from Strathcona Sound and Admiralty Inlet (reference area), Wagemann (1985) observed that nearly half of all the animals from the Strathcona Sound area had significantly elevated levels of at least one of the metals (mercury, lead, cadmium) in either liver or kidney, relative to the reference area. A one-year old seal taken at one of the sampling sites closest to the Nanisivik mine site had abnormally high concentrations (for its age) of both cadmium and lead in the kidney.

2.4 Discussion and Recommendations

Historically, Twin Lakes Creek has been contaminated with lead, zinc, cadmium and other metals as a result of the discharge of effluent from the Nanisivik tailings management area, as well as seepage of acidic, metal-bearing waters from waste rock and exposed outcrops downstream from the tailings. As a result, Twin Lakes Creek has been a significant source of lead, zinc, and cadmium for Strathcona Sound. Elevated metal concentrations, particularly lead, zinc and cadmium, in surface water and sediment, as well as the tissue of invertebrates and macrophytes, have been reported in samples collected in the vicinity of the mouth of Twin Lakes Creek.

Elevated metal concentrations in the tissue of organisms samples indicates exposure to metals, but the extent of negative effects on organisms as a result of exposure to metals is unclear. Only Thomas *et al.* (1983) have examined the health of aquatic organisms in the vicinity of Twin Lakes Creek, and they did not report any negative effects which could be attributed to metals. However, one study of this issue cannot be regarded as decisive.

The current state of the environment in Strathcona Sound is not known. Data more recent that 1983 were not available for the preparation of this case study. This, all effects reported must be regarded as historical. In addition, only water quality monitoring has been conducted in Twin Lakes Creek. The status of biological communities in the creek is not known.

As with many environmental monitoring programs that evolve over time, the Nanisivik Mine monitoring programs would benefit from developing and focusing on specific questions related to aquatic resource protection. Nowhere in any of the documents was a testable hypothesis developed which would focus the studies and specifically test whether or not there were impacts on the aquatic environment receiving mine effluent. If a specific hypothesis or objective is not stated, then the selection of a reference site becomes difficult since it is not clear what parameters are actually being compared. To test for an impact, reference stations should have no exposure to effluent, but be as similar to the exposed locations as possible with respect to "natural" environmental variables.

In response to concerns of the local community about dust blowing from the tailings, road, and ore loading facility, it is recommended that monitoring be conducted to determine to extent of impacts, if any, or organisms important to the local community as food sources. Results of this monitoring should be provided to the community of Arctic Bay. In addition, it is recommended that the Environmental Code of Practice for Mines be revised to include a section dealing with the prevention and control of dust from mine sites.

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Prepared for:
AQUAMIN Steering Group

By:
AQUAMIN Working Group 5
AQUAMIN Secretariat

1996

Working Document

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1 Cluff Lake Mine

1.1 Mine Site Information

1.1.1 Site Setting and Geology

Cluff Lake is in northwestern Saskatchewan, 150 km south of Uranium City and approximately 30 km east of the Alberta border (58°22'N 190°31'W) (Figure 1.1). The significant water bodies in the vicinity of the mine-mill complex are Cluff Lake and its related drainage system, the Douglas River, Snake-Island Lakes drainage system and Sandy Lake. Cluff Lake drains southwestward into the Douglas River via Cluff Creek and a series of small lakes. The Snake-Island Lake drainage system is separate from the Cluff Lake system, but runs in the same southwestward direction to the Douglas River (Figure 1.2).

The Cluff Lake region is part of the Athabasca Plains of the Canadian Shield and is characterized by rolling terrain, sand plains, dunes, moraines, sandy till and bedrock. The uranium deposits are located in the Carswell structure, a geological anomaly lying between Carswell Lake and Cluff Lake in the Athabasca Sandstone Formation (Tones, 1979). This structure consists of an outer ring of highly folded dolomite rocks surrounding an inner circular core of Precambrian metamorphic rocks approximately 20 km in diameter. Glacial activity has sculpted the major landforms in the area.

1.1.2 Mining Operations

Four ore bodies have been mined in the Cluff Lake area:

- D: open pit (1980-82)
- Claude: open pit (1982-89)
- Dominique-Janine; open pit (1990-91)
- Dominique-Peter; underground (1985-present)
- Dominique-Janine Extension; open pit (1994-present)
- Dominique-Janine; underground (1995-present)

Figure 1.1: Location of the Cluff Lake Mine

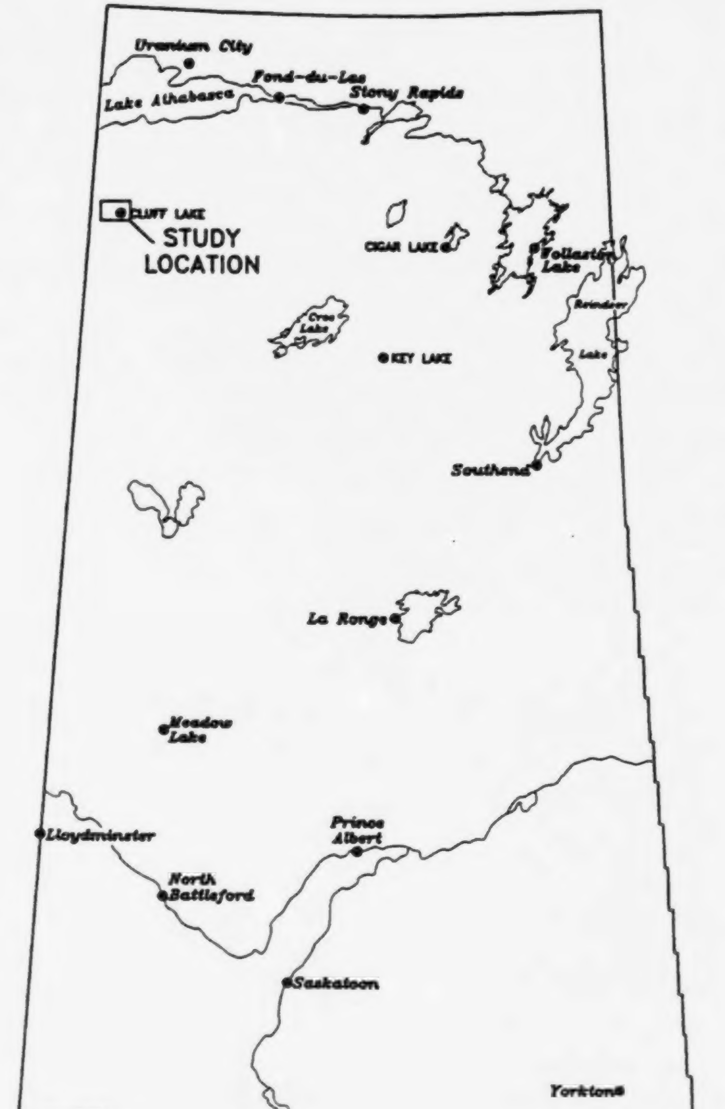
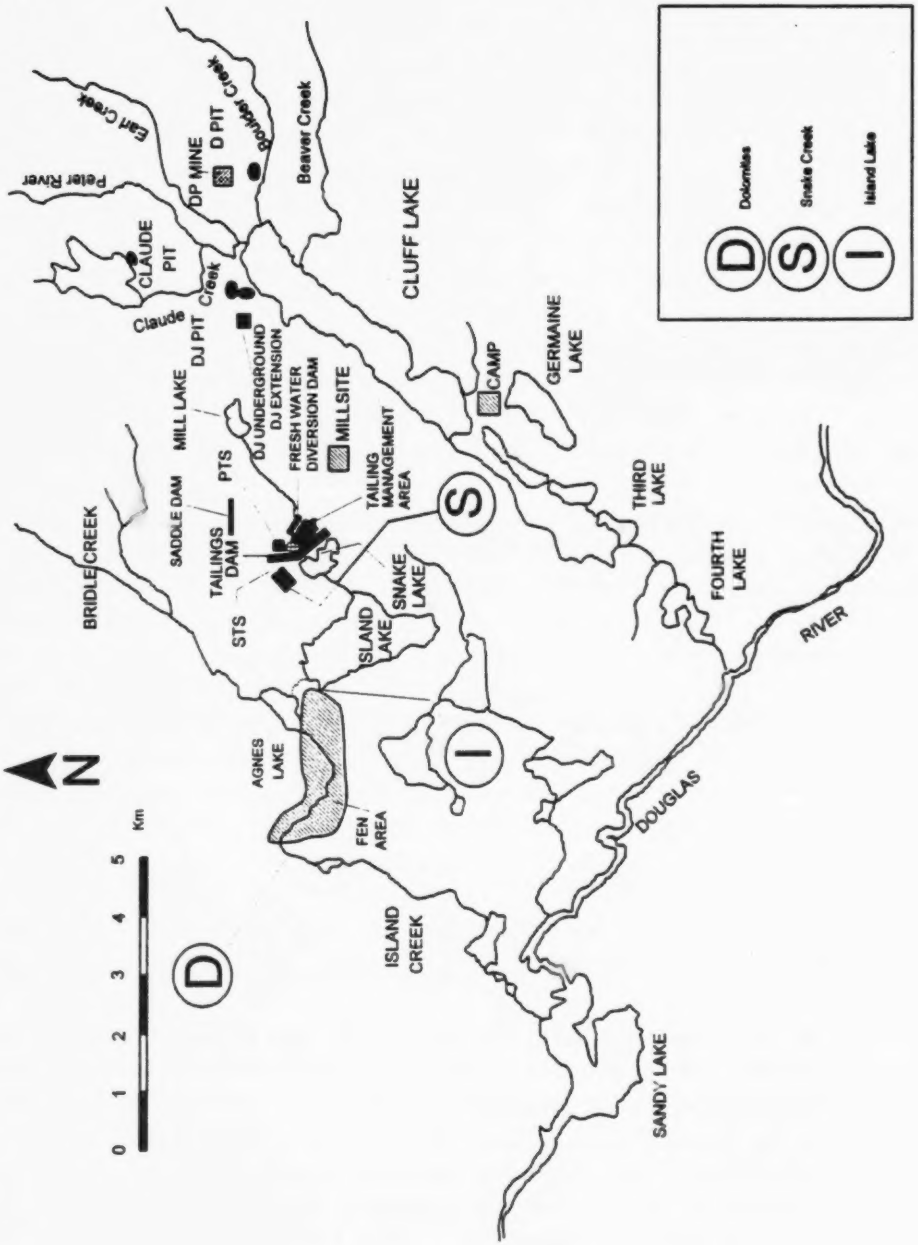

Uranium City
Fond-du-Lac
Lake Athabasca
Stony Rapids
CLUFF LAKE
STUDY LOCATION
CIGAR LAKE
Wollaston Lake
Reindeer Lake
Cree Lake
KEY LAKE
Southend
La Ronge
Meadow Lake
Lloydminster
Prince Albert
North Battleford
Saskatoon
Yorkton
Regina
Maple Creek
Estevan

Figure 1.2: Map of Cluff Lake Mine site and drainage systems


Earl Creek
Peter River
D PIT
DP MINE
Boulder Creek
Beaver Creek
CLAUDE PIT
Claude Creek
DJ PIT
CLUFF LAKE
DJ UNDERGROUND
DJ EXTENSION
FRESH WATER DIVERSION DAM
MILLSITE
MILL LAKE
CAMP
GERMAINE LAKE
PTS
SADDLE DAM
TAILING MANAGEMENT AREA
TAILINGS DAM
STS
S
SNAKE LAKE
ISLAND LAKE
BRIDLE CREEK
THIRD LAKE
FOURTH LAKE
RIVER
AGNES LAKE
I
FEN AREA
DOUGLAS
N
D
0 1 2 3 4 5 Km
ISLAND CREEK
SANDY LAKE
D Dolomites
S Snake Creek
I Island Lake

1.1.3 Mill History

In 1967, Compagnie de Mokta, an original shareholder of Amok Ltd. (now COGEMA Resources Inc.) started exploring the Carswell structure located in north western Saskatchewan. In 1976, the company received its first mineral lease from the Province of Saskatchewan. In June of 1978, approvals for Phase I of the Cluff Lake uranium mining project were obtained. Since start up of operations in 1980, the mill has modified processes three times. During Phase I (May 1980 - Feb 1982), the mine processed primarily high and medium grade ores from the D ore body. Three kinds of tails were produced during Phase I milling processes:

- gravimetric
- leach
- conventional.

Gravimetric tails contained significant quantities of uranium and were stockpiled as a possible feed stock for the facility after completion of Phase I. Leach tails contained significant quantities of gold and radium and were considered unsuitable for disposal in a conventional tailings management facility. They were stored in concrete vaults. Conventional tails were deposited in the Tailings Management Area (TMA).

The Phase I extension mill operated between March 1983 and July 1984 and processed gravimetric tails from the Phase I mill. A grinding facility, a temporary leaching facility and a small extraction facility were constructed during the Phase I extension.

The Phase II mill has been operating since August 1984 and has processed lower ore grades from the other ore bodies.

In 1987 and 1988, a leach tails reprocessing facility was operated to recover gold using cyanide.

1.1.4 Milling Methods (Phase II Mill)

The Phase II mill processing operation consists of primary and secondary crushing, two-stage grinding, acid leaching, counter-current decantation, solvent extraction (kerosene), slurry neutralization, and yellowcake precipitation and drying.

The washed residue from the counter-current decantation circuit is neutralized with milk of lime in three agitated wood stave tanks to a pH of 8.0. Meanwhile, all liquid tailings from other circuits are combined and adjusted to a pH of 8.0 through a series of four agitated solution neutralization tanks. Both final tailings are pumped separately into a tailings pond.

1.1.5 Tailings Management

The tailings management area (TMA) is located in the headwaters of the Mill Creek sub-basin above a tailings dam at the head of Snake Lake (Figure 1.2). Clean water is intercepted above the TMA by a freshwater diversion dam and pumped to Snake Lake on the downstream side of the main tailings dam. The site was selected because it is hydrologically isolated from Cluff Lake and all surface and groundwater from the mill site flow into the facility.

Mine wastes entering the TMA consist of tailings slurry at an average content of 50% solids and liquid tailings with raffinate. Raffinate is a waste stream from the solvent extraction circuit containing some organic solvents and other chemicals. In addition, thickened tailings are deposited intermittently into the TMA, at a projected maximum density of 61% solids.

In 1984, a till dyke was constructed across the tailings pond to intersect the dam and divide the pond into a solids pond and a liquids pond. At the same time, the primary treatment system (PTS) was constructed to treat and settle out chemical precipitates prior to discharge of water into the liquids pond.

In 1986, a till berm was constructed parallel to the dyke to create an upper solids pond into which the enriched tailings, created during leach tails processing (1987, 1988) were placed. This berm is now buried with Phase II tailings.

1.1.6 Effluent Treatment and Discharge

The decant from the solids pond and liquid tailings are directed to the PTS and secondary treatment system, STS (Figure 1.2). Chemical treatment in both the PTS and STS consists of ferric sulphate and barium chloride addition to remove radium. Both facilities can adjust pH of the effluent using soda ash if necessary.

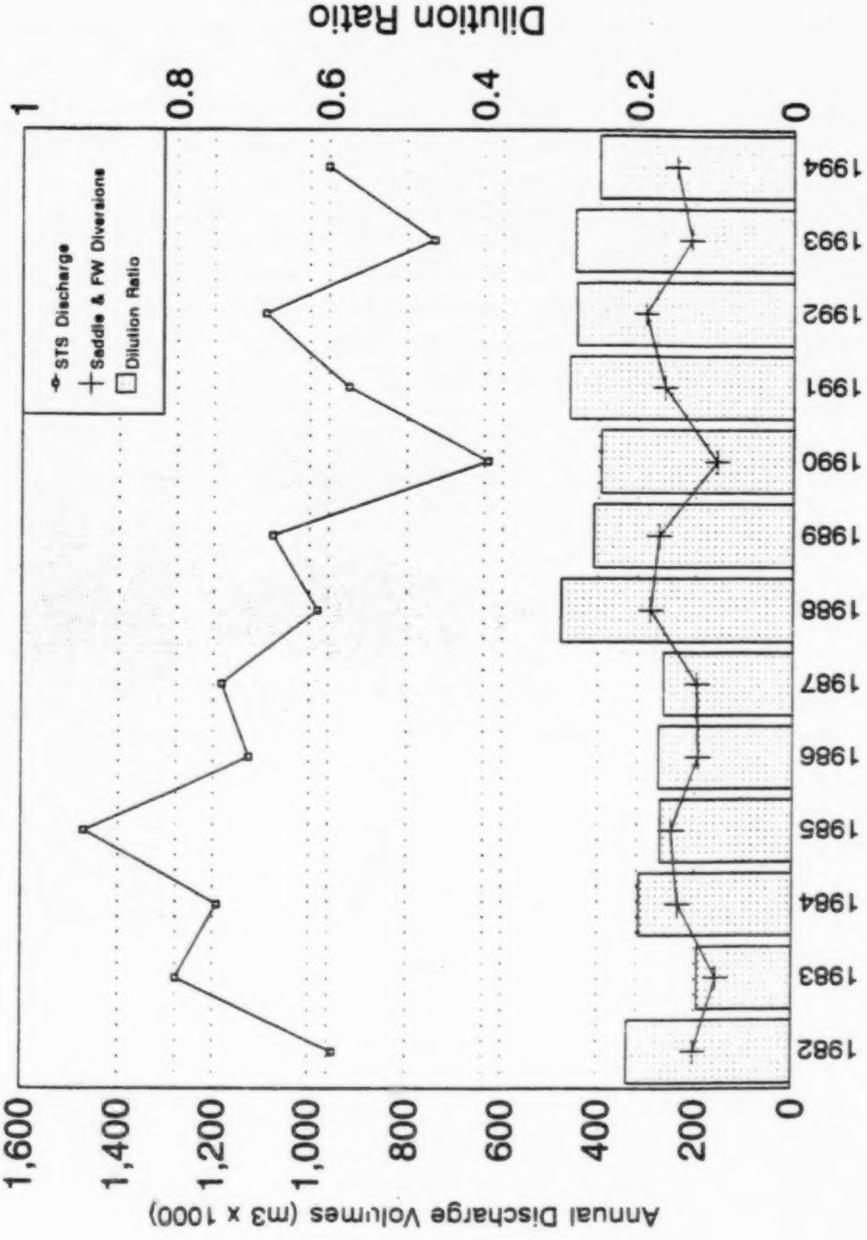
Discharge from the TMA occurs primarily through the STS to Snake Creek. There is some seepage through the dam into Snake Lake at a rate of about 30,000 m^3 per year (Cogema 1993a).

Table 1.1 shows the annual total discharge volume from the STS to Snake Creek and the total annual discharge of Snake Creek above this point. The later is approximated by the annual total water volume diverted from above the Saddle Dam and Freshwater Diversion Dam to Snake Lake. Ratios of these two volumes show that effluent dilution has been very low ranging from a minimum 0.12 in 1983, to a maximum of 0.30 in 1988 (Figure 1.3).

Table 1.1: Cluff Lake operations discharge volumes to Snake Creek

Year	Annual total volume discharge from STS	Annual total volume diverted from Saddle and Fresh Water Diversion Dam (FWDD)	Ratio of Saddle Dam, FWDD : STS
1982	949,853	201,277	0.21
1983	1,280,185	155,251	0.12
1984	1,191,328	235,168	0.20
1985	1,471,138	250,051	0.17
1986	1,124,139	193,628	0.17
1987	1,183,079	196,107	0.17
1988	982,100	292,829	0.30
1989	1,076,326	275,807	0.26
1990	631,267	156,048	0.25
1991	919,602	264,610	0.29
1992	1,092,080	304,318	0.28
1993	744,126	208,911	0.28
1994	961,648	239,888	0.25

Figure 1.3: Cluff Lake Mine effluent discharge volumes to Snake Creek



1.1.7 Compliance Record

1.1.7.1 Secondary Treatment System Effluent

Cogema Resources Inc. has complied with the *Metal Mining Liquid Effluent Regulations* as well as provincial license requirements. All radionuclides and metals, as well as Total Suspended Solids and pH have been within or well below limits given in (Table 1.2) (see Appendix 1 for parameters and sampling frequency; see Appendix 2 for STS Effluent Quality data).

Total ammonia is included in the effluent monitoring requirements, and all compliance monitoring has been conducted as per the site licenses. However, a lack of a consistent data record of total ammonia, and field pH and temperature preclude a full interpretation of these results. Hydroqual (1992) indicated that potential effects of ammonia and uranium on receiving waters warranted further investigation.

Table 1.2: Authorized concentrations of contaminants in liquid effluent

Contaminants	Maximum monthly arithmetic mean concentration	Maximum grab sample concentration
Arsenic	0.5 mg/L	1.0 mg/L
Copper	0.3 mg/L	0.6 mg/L
Lead	0.2 mg/L	0.4 mg/L
Nickel	0.5 mg/L	1.0 mg/L
Selenium	0.6 mg/L	1.2 mg/L
Uranium	2.5 mg/L	5.0 mg/L
Vanadium	0.5 mg/L	1.0 mg/L
Zinc	0.5 mg/L	1.0 mg/L
Ra-226	0.37 Bq/L	1.11 Bq/L
Th-230	1.85 Bq/L	3.7 Bq/L
Pb-210	0.92Bq/L	1.84 Bq/L
Total suspended solids	25.0 mg/L	50 mg/L
Ammonia as NH_3	0.5 mg/L	1.0 mg/L

1.1.7.2 Spills

A variety of spills have occurred on the site since 1990. They are listed in Table 1.3. All spills were contained and treated without any obvious effects on the aquatic system. Most spilled materials were disposed of in the TMA.

Table 1.3: Summary of reportable spills, 1990-1993

Date	Material Spilled	Recovery and Disposal
Jan 3/90	sulphuric acid	neutralized with lime and hauled to TMA
Feb 5/90	sodium chlorate	contaminated ice and snow hauled to mill ore pad contaminate pond for pumping to TMA via mill
May 30/90	diesel fuel	used as starter fluid
Sept 30/90	tailings	liquids and solids moved to TMA
Aug 26/91	hydraulic oil	contaminated soil plus the oil was hauled to the bermed old tank fuel farm disposal area
Dec 28/91	gasoline	contaminated snow moved to a fuel containment berm
Feb 1/92	kerosene	contaminated snow and ice hauled to old tank farm disposal area
Feb 10/92	weak acid water	contaminated material hauled to TMA
Feb 23/92	treated tailings water	contaminated material returned to tailings pond
Mar 1/92	underground slimes	contaminated material hauled to TMA
June 13/92	weak acid solution	hauled to TMA
Oct 3/92	diesel fuel	contaminated material hauled to the old tank farm

1.1.8 Acid Mine Drainage

Cogema (1993b) discussed the potential for acid generation in relation to the extension of the Dominique-Janine ore body. Analysis of core samples suggested that there was potential for acid generation within the extension. Minor but measurable indications of AMD have been recorded and are discussed under water quality in section 1.2.

1.2 Results of Water Quality Monitoring

The current Saskatchewan license requires monitoring of surface water quality at those locations and frequencies listed in Appendix 1. Major ions (HCO_3, CO_3, Cl, Ca, Al, Mg, K, Na SO_4), nutrients (NH_3-N, total nitrogen, total P, TSS), radionuclides (Pb-210, Ra-226, Po-210, Th-230, U), heavy metals (As, B, Hg, Se, Be, Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb, Zn, Ba, Mo, V) and physical properties (pH, conductivity, total hardness, sum of ions, turbidity) are monitored at most sites.

Surface water quality monitoring is also conducted to assess the effects of sanitary sewage on Germaine Lake and other data have been collected for Boulder Creek, and on a one time basis at Agnes Lake Outlet on Island Creek, Island Creek 300 m upstream of Sandy Lake, and at Diane Lake outlet and Bridle Creek (TAEM and SENES 1995). Some data have been collected for the First to Fourth Lakes by TAEM (1993).

The main impacts have been, and were predicted to be, in Island Lake which receives effluent from the TMA. The potential movement of contaminants into the Douglas River system is being monitored.

Prior to the 1995 Cogema Resources Cluff Lake Operations Status of the Environment Report, data analyses consisted primarily of graphical presentations to highlight changes from year to year at specific locations, and changes from place to place within specific years. For some of parameters, graphic presentation of trends is adequate because the magnitude of the changes over time or differences from site to site have been substantial (e.g., uranium in the Island Lake drainage). TAEM (1994) reviewed the observed mining related changes in water quality since operation commenced. They concluded that Snake Lake has been affected by seepage from the TMA, the STS and mine construction. They also concluded that discharge from the STS and water from Snake Lake had altered water chemistry in Snake Creek, Island Lake, Agnes Lake, Island Creek, Sandy Lake and the Douglas River. Agnes Lake is likely impacted due to the recent formation of a channel that connects Agnes Lake to Island Lake. Higher nutrient levels in Island Lake are thought to have contributed to bog/cattail growth in outlet areas of Island Lake resulting in a lake-level increase. The lake-level increase has caused water to start flowing from Island Lake out to Agnes Lake through a low area (L. Sinclair, pers. comm., 1995). The fen downstream of Island Lake is thought to be responsible for major reductions in concentrations of metals and radionuclides in the system downstream of Island Lake (Figure 1.2).

1.2.1 Snake Lake

The main changes in water quality have been increases in major ions, conductivity and total dissolved solids (Figure 1.4). Uranium concentrations in water are an order of magnitude higher than baseline levels (Figure 1.5), however these values remain two orders of magnitude lower than Saskatchewan Municipal Drinking Water Quality Objectives and well below the observed no effect level for aquatic organisms (see Table 1.4). No other radionuclides or trace metals were elevated (TAEM, 1994).

TAEM 1995 identified that manganese (during the period 1989 to 1994) peaked in 1990 at 0.12 mg/L and has since decreased to levels observed in 1989 (mean = .045 mg/L).

Figure 1.4: Total dissolved solids, sums of ions, and specific conductivity in Snake Lake, before (base) and after mining commencement of mining operations (after TAEM 1994).

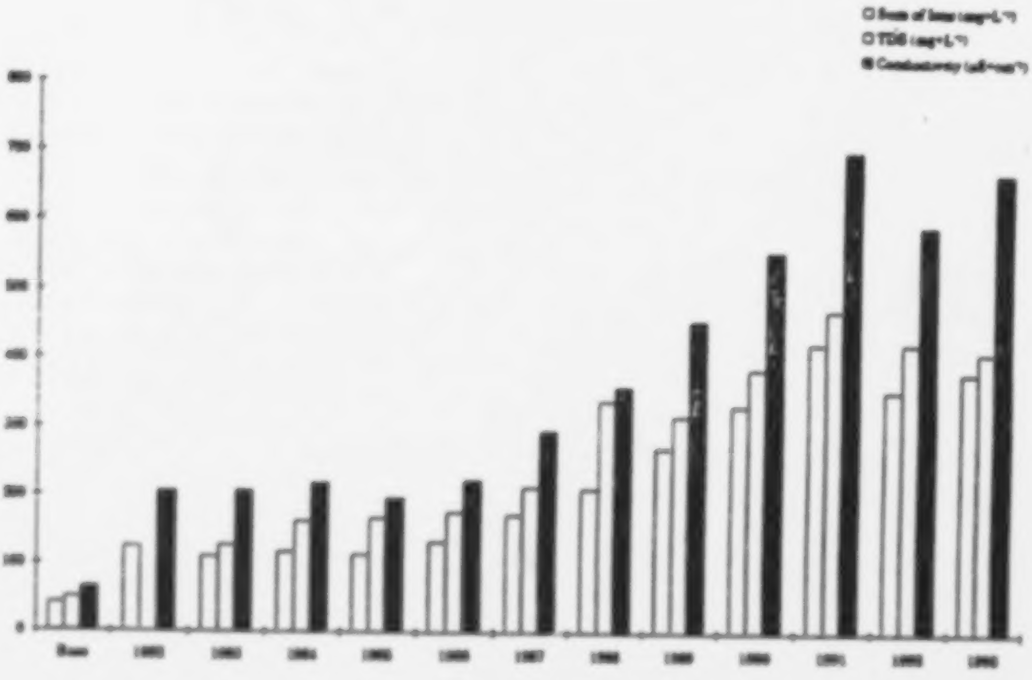
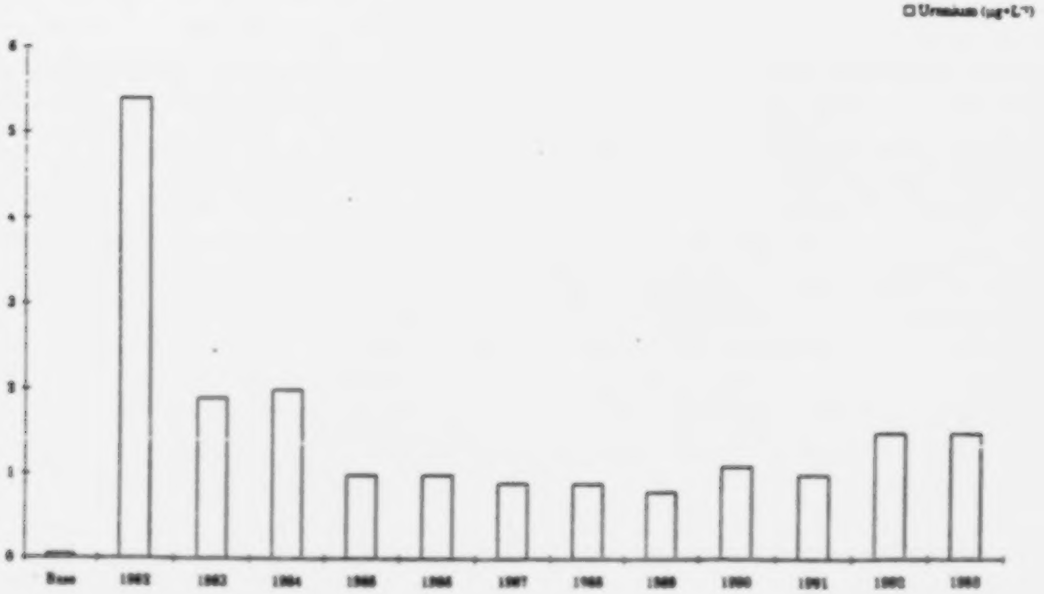


Figure 1.5: Uranium concentrations in Snake Lake, before (base) and after mining commencement of mining operations (after TAEM 1994).



1.2.2 Island Lake

Water quality of Island Lake has changed more than any other component of the watershed. In Appendix 3 detailed information on water quality in Island Lake is presented in the form of line graphs of discrete sample data from 1981 baseline to 1995. The data show an upward trend for a number of the parameters measured until about 1989, followed by a decreasing trend. This gradual improvement in water quality since 1989 corresponds with increased dilution of STS effluent by Snake Creek (i.e. Freshwater Diversion Dam and Saddle Dam Discharge) after 1987 from about 0.17 to between 0.25 to 0.30 (see Table 1.1 and Figure 1.3).

Total dissolved solids reflect the main trend and increased to levels up to 2700 mg/L (specific conductivity 4,500 µS/cm) from a baseline of 77 mg/L, then decreased by 1994 to below a specific conductivity of 3,500 µS/cm (see Figure 1.6). TAEM (1994) suggest that the lake has approached saline status. Sulphate primarily, but also calcium, magnesium, potassium and sodium are responsible for increased salinity (Figures 1.6 and 1.7).

Figure 1.6: Total dissolved solids, sulphate, and specific conductivity in Island Lake, before (base) and after mining commencement of mining operations (after TAEM 1994).

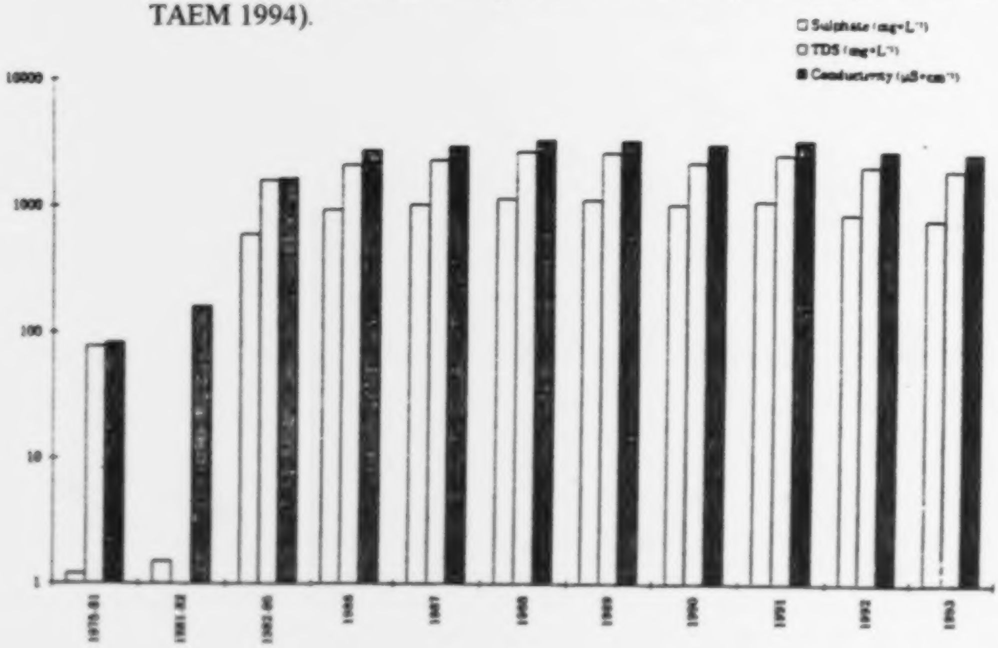
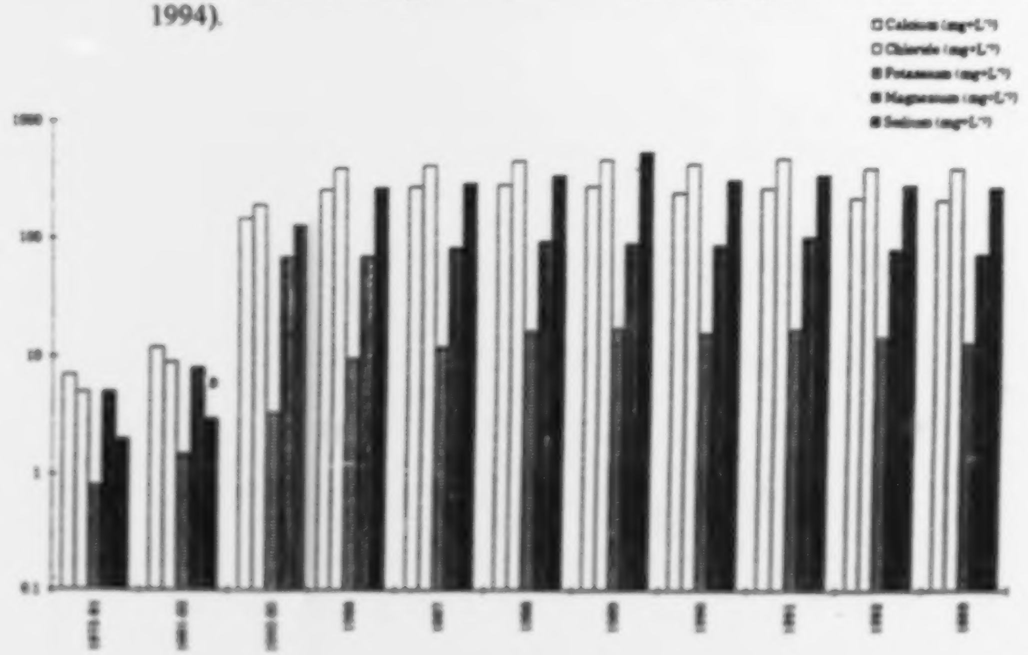


Figure 1.7: Calcium, chloride, potassium, magnesium, and sodium in Island Lake, before (base) and after mining commencement of mining operations (after TAEM 1994).



High salinity is detrimental to most forms of freshwater life, but sensitivity to salinity often spans two orders of magnitude within any plant or animal taxa. Even closely related species from freshwater show surprisingly wide ranges in tolerance, and may exhibit acclimation to increased salinity. Consequently, it is difficult to specify a single toxic limit for salinity. Long term changes in the flora and fauna of natural lakes begin to appear when salinity exceeds about 1000 mg/L; this level is widely used as the dividing line between freshwater and saline lakes (Hammer and Haseltine 1988). Generally, total ionic concentration rather than ionic composition determines the biological effect of saline water (NRCC 1977), however individual ions are known to be toxic in concentrated solutions. For example, the safe level of potassium in water for the survival of freshwater mussels is predicted to be 4-10 mg/L (Imlay 1973).

In Island Lake concentrations of arsenic, copper, nickel and vanadium showed either no apparent increases, or minor increases. In all cases, concentrations remained far below Saskatchewan Surface Water Quality Objectives for the protection of aquatic life and wildlife (SSWQO 1988 - Appendix 4). Selenium concentrations have generally remained below the SSWQO of 0.021 mg/L and show no trend.

Of the metals that have been monitored, only molybdenum has exhibited an increase in Island Lake. By 1986, molybdenum concentrations had increased two orders of magnitude (5 to 540 μg/L). Since 1986, concentrations have remained near 570 μg/L (see Figure 1.8). The concentration at which molybdenum can have toxic effects on aquatic organisms appears to be high (e.g. 96 h LC$_{50}$ for rainbow trout 800 to 7340 mg/L, Eisler 1989, Stephenson *et al.* 1991). However, the sublethal effects of molybdenum on other biota such as phytoplankton could have significant ecosystem effects. A literature review should be conducted to determine potential effects.

Concentrations of the radionuclides ^{230}thorium, ^{226}radium, ^{210}lead and ^{210} poloniuim have remained at or near baseline levels except for a few peaks generally in the early years of operation. However, uranium has gone through a 3 order of magnitude increase in concentration in Island Lake from less than 1 μg/L in 1981 to 2900 μg/L in 1985 (see Figure 1.9). Following this peak concentration, uranium concentrations have steadily decreased to a seasonal range of 110 to 240 μg/L in 1994.

The range of uranium concentrations spanned during operations has exceeded levels known to be toxic to aquatic life seen in Table 1.4. Current ranges approximate the lowest observable effect concentrations for a number of the most sensitive species or life stages of species. This includes reduced larval fish growth at 200 μg/L (Holdway 1992), and a lowest observed effect concentration (LOEC) for Hydra reproduction at 190 μg/L (Allison and Holdway 1988).

Figure 1.8: Molybdenum concentrations in Island Lake from baseline studies to 1993 (after TAEM 1994).

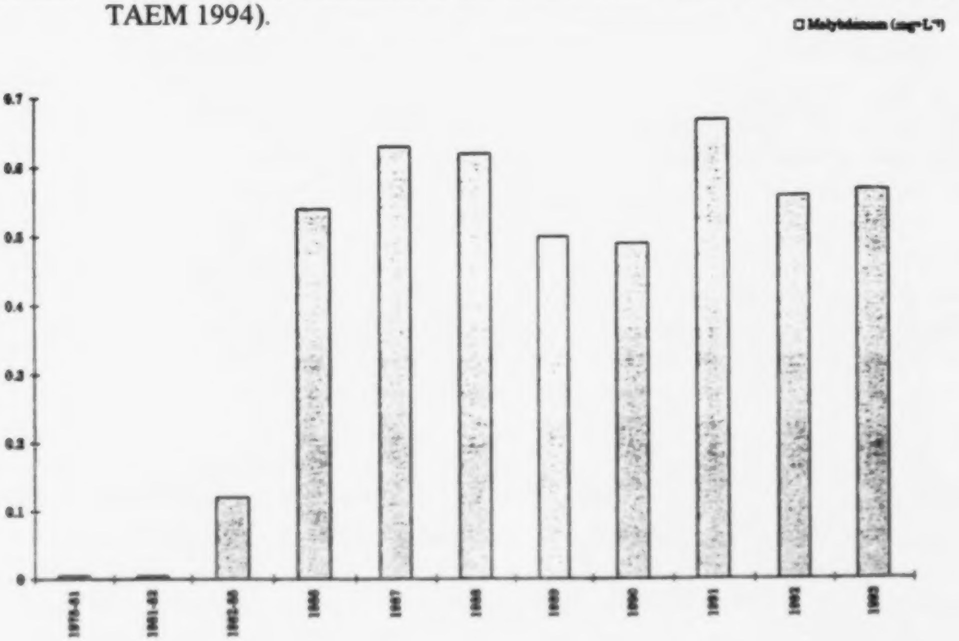
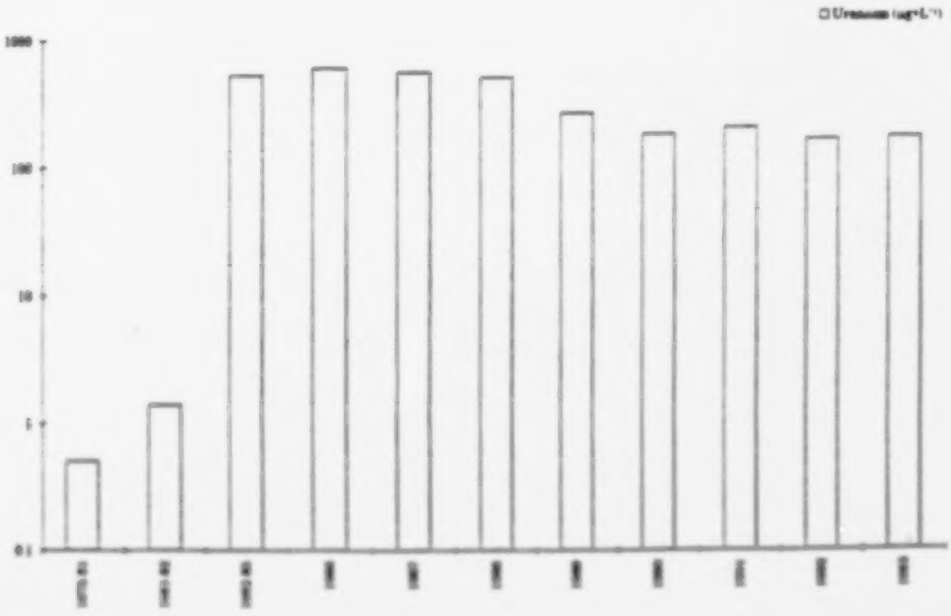


Figure 1.9: Uranium concentrations in Island Lake from baseline studies to 1993 (after TAEM 1994).



Uranium toxicity spans a wide range of concentrations and is species/taxa dependent. Also, uranium toxicity like other metals is apparently modified by pH and ionic composition. For example, the addition of bicarbonate increased the no observable effect concentration (NOEC) for reproduction of Hydra to 3900 μg/L in uranium mine retention pond water (Table 1.4). Thus, derivation of a release limit must be based on site specific testing of native species (or appropriate surrogates) using a range of expected mixing zone dilution rates in site specific dilution water.

Phosphorous concentrations in Island Lake are elevated but have decreased below 0.1 mg/L since reaching a high mean of 0.22 mg/L in 1982-1985 (Figure 1.10). The 1994-1995 concentrations have apparently returned to the 1988 to 1990 concentrations of about 0.04 mg/L. Lakes with phosphorus concentrations above 0.05 mg/L have high primary production of phytoplankton or aquatic plants and are classified as hypereutrophic (TAEM and SENES 1995).

Ammonia concentrations in the lake continue to be elevated above baseline, but a sparse data set including field pH, temperature and total ammonia ($NH_3 + NH_4$-N) preclude long term trend assessment. A more continuous record of total ammonia, field temperature and pH are needed.

Figure 1.10: Phosphorous concentrations in Island Lake from baseline studies to 1993 (after TAEM 1994).

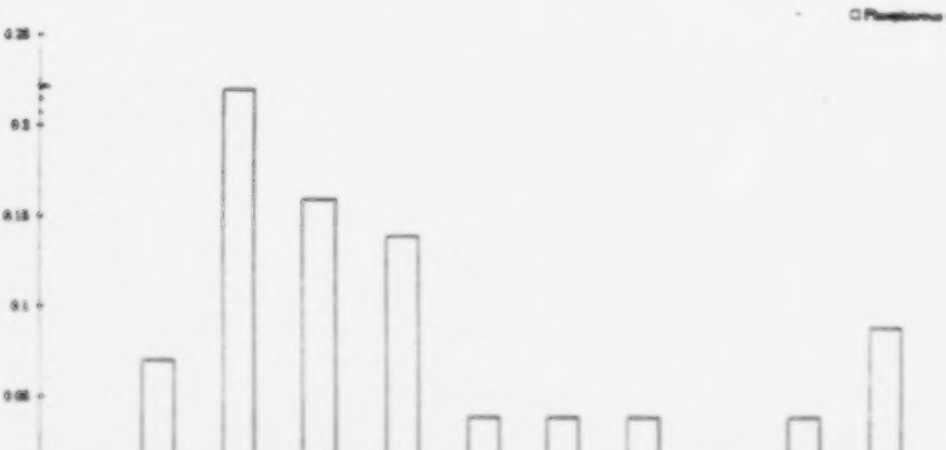


Table 1.4: Toxicity test results and calculated effect safe levels for aquatic organisms exposure to uranium

Concentration	Test/effect level	Species	Note
2800 μg/L (1)	96h LC_{50}	fathead minnow (*Pimephales promelas*)	soft water
6200 μg/L (1)	96h LC_{50}	rainbow trout (*Salmo gairdneri*)	hard water
8000 μg/L (1)	96h LC_{50}	brook trout (*Salvelinus fontinalis*)	hard water
500 μg/L (2)	reduced reproduction	waterflea (*Daphnia magna*)	
190 μg/L (3)	LOEC for reproduction	hydra (*Hydra viridissima*)	
160 μg/L (3)	NOEC for reproduction	hydra (*Hydra viridissima*)	
440 μg/L (4)	NOEC based on 14 day exposure, 15 day post exposure observation of one day old larval fish	purple spotted gudgen (*Morgurnda morgurnda*)	low conductivity 29 μS/cm
<404 μg/L (4)	NOEC based on reduced growth of one day old larval fish	purple spotted gudgen (*Morgurnda morgurnda*)	low conductivity 29 μS/cm
1570 μg/L (4)	96h LC_{50} 6 day old fish larvae	purple spotted gudgen (*Morgurnda morgurnda*)	low conductivity 29 μS/cm
700 μg/L (4)	96h LC_1 6 day old fish larvae	purple spotted gudgen (*Morgurnda morgurnda*)	low conductivity 29 μS/cm
3290 μg/L (4)	96h LC_{50} 40 and 70 day old fish	purple spotted gudgen (*Morgurnda morgurnda*)	low conductivity 29 μS/cm
200 μg/L (4)	calculated threshold for larval fish growth	purple spotted gudgen (*Morgurnda morgurnda*)	low conductivity 29 μS/cm
20μg/L (4)	calculated safe level for 1 day old fish larvae	purple spotted gudgen (*Morgurnda morgurnda*)	low conductivity 29 μS/cm
70 μg/L (4)	alternate safe level based on LC_1 lower 95% fiducial limit for 1 day old fish larvae based on 14 day exposure and 15 day post exposure observation	purple spotted gudgen (*Morgurnda morgurnda*)	low conductivity 29 μS/cm
200 μg/L (5)	48H LOEC based on survival (fed green algae during test)	water flea (*Moinodaphnia macleavi*)	low conductivity 20 to 27 μS/cm
25 μg/L (5)	5d LOEC based on survival (fed green algae during test)	water flea (*Moinodaphnia macleavi*)	low conductivity 20 to 27 μS/cm
3900 μg/L (5)	NOEC for reproduction	hydra (*Hydra viridissima*)	mine retention pond water at pH 8.5, bicarbonate buffer
900 μg/L (5)	NOEC for reproduction	hydra (*Hydra viridissima*)	pH 8.5 in bicarbonate buffered low conductivity water
≥200 μg/L (5)	LOEC for reproduction	hydra (*Hydra viridissima*)	pH 6.5 in unbuffered water

(1) Davis 1980
(2) Poston *et al.* 1984
(3) Allison and Holdway 1988
(4) Holdway 1992
(5) Hyne *et al.* 1993

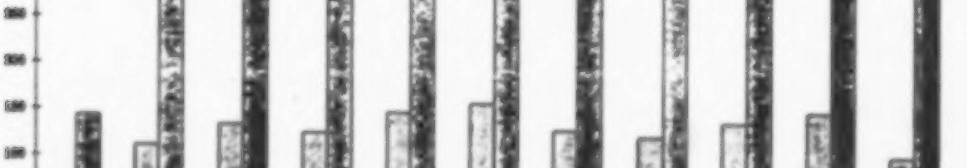
1.2.3 Island Creek at the Dolomites

Successive reports (TAEM 1994 and 1994, Swanson 1991) and data in Appendix 5 confirm that the fen below Island Lake effectively reduces concentrations of most contaminants to below SSWQ objectives (Appendix 4). Since 1981, only zinc concentrations have exceeded the SSWQ of 0.05 mg/L during 1982. This point may be an anomaly.

There are no major trends in concentrations of any of the radionuclides and while uranium concentrations have shown peaks of about 6 to 11 μg/L, they have remained near baseline.

The main long term effect on Island Creek below the fen has been an increase in conductivity from a baseline of about 86 μS/cm to seasonal ranges of about 300 to over 800 μS/cm. These seasonal ranges are also evident for the concentrations of major ions including potassium, magnesium, calcium and sodium (Appendix 5). Sulphate concentrations have also increased from baseline of about 35 mg/L to near 100 mg/L (Figure 1.11). TAEM and SENES (1995) identified a step wise trend in molybdenum concentrations since 1991 from 0.01 mg/L to the 0.027 mg/L range, and a small decrease in alkalinity of 5 to 10 mg/L to about 60 to 63 mg/L. The trend in molybdenum concentrations is shown in Appendix 5. Further, ammonia concentrations were found to be substantially reduced compared to Island Lake, but phosphorous remained near concentrations for Island Lake occasionally peaking into the hypereutrophic zone.

Figure 1.11: Total dissolved solids and sulphate in Island Creek at the Dolomites, from baseline studies to 1993 (after TAEM 1994).



1.2.4 Island Creek at Sandy Lake

Molybdenum concentrations have increased from a baseline of 0.005 to over 0.025 mg/L in 1993 (Figure 1.12). Recently TAEM and SENES (1995) noted statistically significant changes in concentrations of molybdenum, manganese, iron and total nitrogen between 1989 to 1994 for Island Creek at Sandy Lake. Molybdenum concentrations increased from 0.011 to 0.022 mg/L in this period reflecting the increase below the fen station. Also, iron and manganese concentrations peaked in 1993 and decreased to 1989 levels in 1994. Total nitrogen concentrations have doubled since 1984, but have levelled off. Phosphorous concentrations are similar to those in Island Lake and again peaked into hypereutrophic levels. Uranium concentrations have shown considerable changes since 1981 showing increases up to 5 times baseline up to about 2.8 μg/L (Figure 1.13). However, the later concentration is within the natural range for uranium in Saskatchewan surface water.

Figure 1.12: Molybdenum concentrations in Island Creek at Sandy Lake, from baseline studies to 1993 (after TAEM 1994).

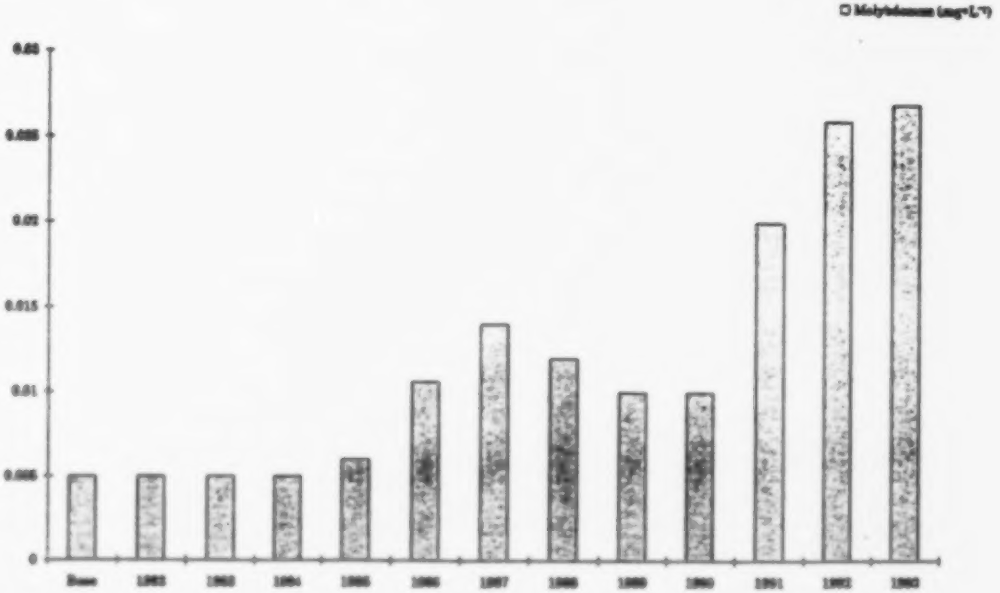
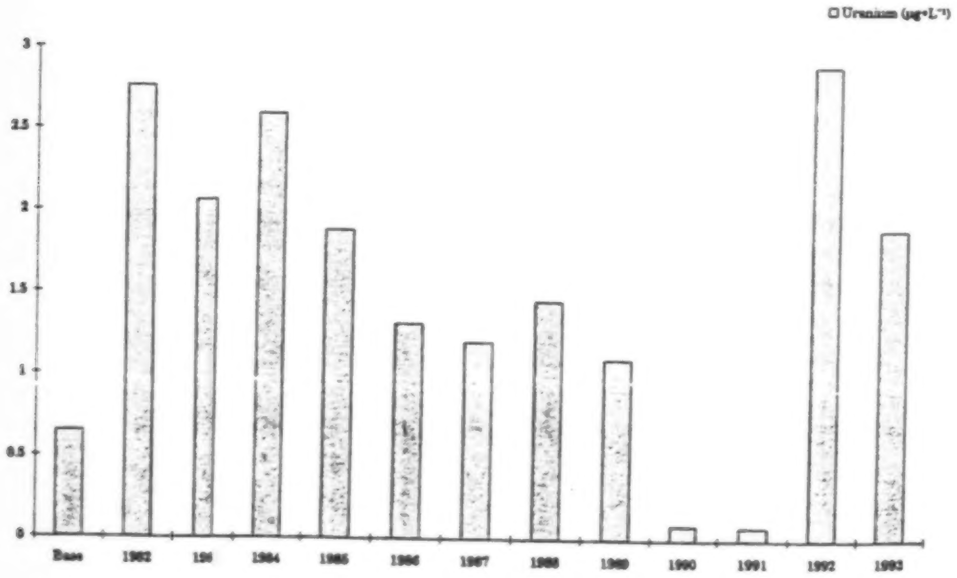


Figure 1.13: Uranium concentrations in Island Creek at Sandy Lake, from baseline studies to 1993 (after TAEM 1994).



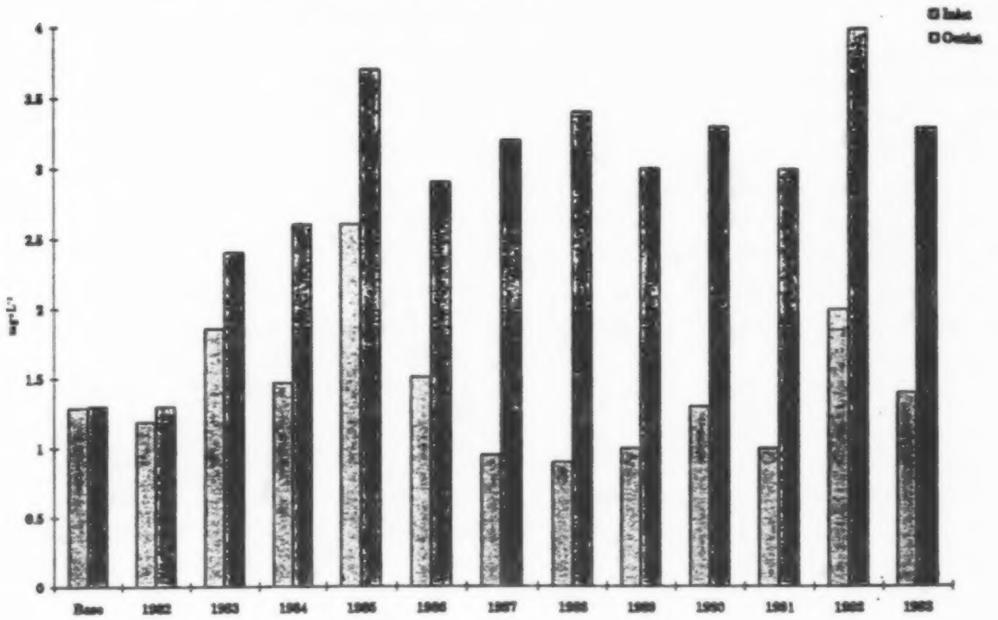
1.2.5 Douglas Rver Inlet and Outlet of Sandy Lake

Water quality statios situated at the inlet and outlet to Sandy Lake on the Douglas River system serve to detec differences in water quality attributable to Island Creek discharge. At the outlet, sulphate hs doubled from baseline concentrations of about 1.25 mg/L to about 3.25 mg/L in 1993 (igure 1.14).

No trends have been etected in concentrations of uranium or radionuclides at these locations since operations begn.

TAEM and SENES 1995) analysed the differences between mean inlet and outlet water quality for the periodl989 to 1994 and found statistically significant increases at the outlet of Sandy Lake in pHspecific conductance, total alkalinity and hardness, and concentrations of bicarbonate, sulpate, calcium, sodium and sum of ions. They also found decreases in concentrations of TS and iron. While measurable, these changes were considered to be small and will not reult in impairment of the Douglas River (see Appendix 6).

Figure 1.14: Sulphate concentrations at the outlet and inlet of Sandy Lake, from baseline studies to 1993 (after TAEM 1994).



1.2.6 Agnes Lake

The initial flow path from Island Lake to Island Creek progressed mainly through a direct outlet into Island Creek. This outlet has since developed into a fen with limited flow capacity, and the water level in Island Lake has risen as a result. Within the last few years another channel within the same area has opened up, and the effluent now proceeds into Agnes Lake, en route to Island Creek. Although sampling has started at Agnes Lake, insufficient data exist for assessment of effects.

1.2.7 Cluff Lake Drainage

Claude Creek, Peter River, Earl Creek and Boulder Creek feed Cluff Lake which flows out through Cluff Creek by Third and Fourth Lakes to the Douglas River. Third Lake also receives input from First and Second Lakes, which are not part of the Cluff Lake drainage.

According to Swanson (1990), there is little evidence of substantial changes from baseline concentrations of any of the water quality parameters. In part, the lack of detection of changes was attributed to having too few stations on any given stream to evaluate potential location-specific impacts. A small increase in suspended solids was detected in the Peter River at the Mill Road in 1988. For Earl Creek, stations above and below the cement batch plant have shown increased hardness, alkalinity, conductivity, pH, and concentrations of TDS, calcium, magnesium and bicarbonate. These changes may be due to the cement batch plant operations (L. Sinclair pers. comm.) between 1989 to 1994 (TAEM and SENES 1995). Water samples from the Douglas River system above and below Cluff Creek show no significant increases in any water quality parameters.

In reviewing data for 1989 to 1994 TAEM and SENES (1995) noted only one measurement above the SSWQ objective for zinc of 0.05 mg/L at 0.087 mg/L in Boulder Creek. In 1993 an increase in sulphate concentration was detected (6.1 mg/L versus 1.6 to 2.2 mg in previous years). Small increases in pH, and concentrations of carbonate, magnesium, and sodium were also noted.

For the Peter River, TAEM and SENES (1995) noted only one exceedence of the SSWQO for zinc at 0.08 mg/L, versus the 0.05 mg/L objective. Sulphate concentrations have increased from a background of about 1.25 mg/L to 7.5 mg/L in 1994 suggesting a minor but measurable increase from acidic drainage of waste rock piles. Uranium was detected in 1992 and 1993 at about 1.25 mg/L but returned to baseline at 0.5 mg/L in 1994. Small increases in turbidity and decreases in alkalinity and bicarbonate concentration were noted. TAEM and SENES (1995) indicate that water quality is similar to Swanson (1991) who reported no significant changes from baseline. However, on one occasion in 1994 pH was 6.17, lower than the SSWQO of 6.5, and there has been an 8 fold increase in mean iron concentrations from a baseline level of about 0.85 mg/L in 1989 to 6.93 in 1994. This exceeds the SSWQO of 1 mg/L. No significant increases in conductivity, turbidity, TDS, alkalinity, and bicarbonate were noted.

Cluff Lake data in TAEM and SENES (1995) show trends in mean annual concentrations of sulphate from 1.8 (1989) to 4 mg/L (1994) and uranium from 0.11 (1989) to 1.74 mg/L (1994) and a decreasing trend in pH from 7.84 (1989) to 7.75 (1994). These changes in water quality may be indicative of inputs from acidic drainage from waste rockpiles in the upper drainage. Other than these changes, concentrations of other parameters including arsenic, molybdenum, nickel, radium 226, and TDS are similar to baseline (TAEM and SENES 1995). No changes have been detected to date in the First to Fourth Lakes.

Germaine Lake receives seepage from a shoreline tilefield. Treated effluent is pumped from the Germaine Camp sewage treatment plant. In 1986, it was reported that the sanitary sewage seepage into Germaine Lake was about 0.8% of the volume of the lake. Water quality data from Germaine Lake in 1985 and 1986 showed very low nutrient content, low BOD, low TSS and low specific conductivity. There are no baseline data for the lake against which to evaluate the significance of the observed conditions.

1.2.8 Summary and Recommendations

1. Impacts on water quality have occurred in Snake Creek and in Island Lake. Minor effects have been detected in lower Island Creek, Sandy Lake, on the Douglas River System, and in the upper drainage of the Cluff system including Cluff Lake.

2. Total ammonia data are lacking for the Island Creek drainage. These data along with pH and temperature are needed to adequately assess potential un-ionized ammonia toxicity.

3. Salinity and uranium should be considered for inclusion in the revised MMLER, recognising that they are specific to the uranium industry, and may not meet criteria for inclusion in a national regulation.

4. Molybdenum and ammonia, which are not specific to the uranium industry, should be considered for inclusion in the revised MMLER.

5. A method of setting limits based on site specific receiving water dilution (or loadings) is needed.

1.3 Results of Sediment Quality Monitoring

TAEM and SENES (1995) have summarized baseline and operational sediment quality data collected since 1975 in the Island and Cluff Lake drainage. Sediments were also collected in the Saskatchewan Environment and Resource Management, Cumulative Effects Monitoring Program (SERM CEM) on the Douglas River system at three locations in 1995[1]. A summary of the sampling programs is presented in Table 1.5.

The current monitoring requirements for the mine site include collection of samples bi-annually at Snake Lake near the outlet, Island Lake near the outlet and Cluff Lake near the outlet. Concentrations of Al, As, Cu, Mo, Ni, Pb, Zn, Pb-210, Po-210, Ra-226, Th-230 and U are measured, as are loss on ignition (LOI, % organic matter) and % moisture.

Baseline data were collected from Mill Lake, Snake Lake, Island Lake, 2 lakes on upper Bridle Creek, Sandy Lake, Peter River Outlet, a lake at upper end of Peter River, Claude Lake and next lake upstream, Boulder Creek at Cluff Lake, Lake at Upper Boulder Creek, Cluff Lake north and south, Germaine Lake, First to Fourth Lakes, Cluff Creek near confluence and the Douglas River.

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[1] Note that, at the time of preparation of this case study, SERM CEM results were draft, and had not been reviewed by Cogema Resources

Operational sampling has also been done at a number of the locations. In 1985, Hynes completed an extensive survey of sediments in Island Lake.

Baseline samples were collected using grab methods such as the Ekman grab which are known to inaccurately sample light flocculant surficial layers because of bow waves. Following baseline collection some samples have been collected using impact corers which more accurately determine depth stratification of chemicals, but which also subject surficial strata to a bow wave before the sample is collected. Given the changes in sampling technique, only rough comparisons to baseline condition can be made using surficial strata data. However, the comparisons have been sufficient to detect general trends in sediment quality.

1.3.1 Snake Lake and Island Lake Drainage

Snake Lake outlet sediments (0-10 cm) showed increases in concentrations of copper, molybdenum, nickel and uranium between 1978 to 1993 (Table 1.6). Copper increased slightly, from 7.6 to 9, while nickel increased from 1.33 to 16 μg/g, molybdenum increased 4 times from 2.7 to 11 ug/g and uranium increased from 0.4 ug/g to 8.4 ug/g, a 21 fold increase.

For Island Lake near the outlet, major increases between 1978 to 1993 in the 0-10 cm strata were recorded for concentrations of molybdenum (3.5 to 160 ug/g), and uranium (0.4 to 102 ug/g). Minor increases were recorded for concentrations of nickel (11.6 to 14 μg/g) (Table 1.7).

Data for uranium at the two locations above show considerable variations which may indicate poor year to year return to site accuracy for sediment collection or some other artifact of sampling error. A decreasing trend from a peak concentration in Snake Lake for uranium at 24 μg/g in 1991, and 701μg/g in Island Lake demonstrate the variability (Table 1.6 and 1.7).

Assessment of the biological significance of uranium or molybdenum accumulation is not possible since no sediment quality guidelines are available. However, 1993 concentrations of copper and nickel in Snake Lake are less than guideline threshold biological effect levels (TEL)(Smith *et al.* 1995, Appendix 7).

Table 1.5: Sediment sampling locations and year sampled

Location	Year Sampled	Source
Mill Lake	1978	Dunn 1980
Snake Lake	1978, 1980-1987, 1990-1994	Tones 1979; Cluff Environment; TAEM 1994
Island Lake	1978, 1981-1987, 1987, 1990-1993	Tones 1979; Dunn 1980; Cluff Environment; Hynes 1990; TAEM 1994
Island Creek	1978, 1993	Tones 1979; TAEM 1994
Bridle Creek	1993, 1994	TAEM 1993; Cluff Environment
2 lakes on upper Bridle Creek drainage	1978	Dunn 1980
Agnes Lake (2 sites)	1993, 1994	TAEM 1993; Cluff Environment
Sandy Lake (2 sites)	1978	Tones 1979
Peter River Outlet	1978, 1984-1986	Dunn 1980; Cluff Environment
Lake at upper end of Peter River	1978	Dunn 1980
Claude Creek at Claude Lake	1984-1986	Cluff Environment
Claude Lake & next lake upstream	1978	Dunn 1980
Boulder Creek at Cluff Lake	1978, 1980-1986	Tones 1979; Dunn 1980; Cluff Environment
Lake at upper end of Boulder Creek	1978	Dunn 1980
Cluff Lake North	1975, 1978, 1988, 1993	Sawchyn 1975; Tones 1979; Dunn 1980; Swanson 1990; TAEM 1993
Cluff Lake mid-lake	1988, 1993	Swanson 1990; TAEM 1993
Cluff Lake South	1975, 1978, 1988, 1993	Sawchyn 1975; Tones 1979; Dunn 1980; Swanson 1990; TAEM 1993
Cluff Lake Outlet	1984-1986, 1990-1994	Cluff Environment
Germaine Lake	1978	Dunn 1980
First to Fourth Lakes	1978, 1988, 1993	Dunn 1980; Swanson 1990; TAEM 1993
Cluff Creek Near Confluence	1975, 1978	Sawchyn 1975; Tones 1979
Douglas River (2 sites)	1975	Sawchyn 1975
Control Lake on Douglas River about 30k Above Cluff Mine Site	1995	SERM CEM Program
Inlet to Sandy Lake	1995	SERM CEM Program
Delta of Island Creek in Sandy Lake	1995	SERM CEM Program

Table 1.6: Comparison of radionuclide and trace element concentrations in sediments collected near the outlet of Snake Lake, 1978 to 1993 (TAEM 1994)

Year	As	Cu	Mo	Ni	Pb	U	Th	^{210}Pb	^{226}Ra	^{230}Th	Moisture %
	µg/g							Bq/g			
1978	2	7.6	2.7	13.3	8.9	0.4	3.7	0.33	0.12	-	-
1980	-	-	-	-	-	0.8-1.1	1.7-5.4	0.05-3.7	0.03-30	-	-
1981	-	-	-	-	-	2.0-11.9	1.6-3.6	0.20-0.60	0.04-0.08	-	-
1982	-	-	-	-	-	9.5-14.8	0.18-1.8	0.60-0.17	0.04-0.06	-	-
1983	-	-	-	-	-	4.2-11.0	1.2-2.5	0.06-0.12	0.06-0.10	-	-
1984	-	-	-	-	-	5.6-6.4	2.0-2.8	0.08-0.12	0.04-0.07	-	-
1985	0.82	3.9	< 4	38	4	5.7	2.8	0.11	0.07	-	-
1986	1.6	< 2	< 5	6	< 5	5.6	-	-	0.06	-	-
1987	0.3	2.8	5	5	12	2.2	-	0.03	0.02	0.03	38.6
1990	0.4	1	4	2	7	1.3	-	0.13	0.02	0.03	37.9
1991	1.8	5	12	8	5	24	-	0.51	0.11	0.06	84.7
1992	2.1	7	9	8	27	17	-	0.3	0.1	0.09	86.3
1993*	2.1	9	11	16	< 4	8.4	-	0.14	0.13	0.05	90.8

* 1993 results based on top 10 cm of core samples; all other years based on surface grab samples using Ekman dredge.

Table 1.7: Comparison of radionuclide and trace element concentrations in sediments collected near the outlet of Island Lake, 1978 to 1993 (TAEM 1994)

Year	As	Cu	Mo	Ni	Pb	U	Th	^{210}Pb	^{226}Ra	^{230}Th	Moisture %
	µg/g							Bq/g			
1978	2.1	7.5	3.5	11.6	4.9	0.4	4	0.12	0.12	-	-
1981	-	-	-	-	-	3.0-7.7	0.25-4.0	0.045-0.8	0.01-0.15	-	-
1982	-	-	-	-	-	17.5-217	<0.1-0.5	0.03-1.1	<0.005-0.16	-	-
1983	-	-	-	-	-	10.2-189	1.1-1.4	0.12-0.8	0.05-0.2	-	-
1984	-	-	-	-	-	48-185	1.4-2.0	0.16-0.5	0.05-0.25	-	-
1985	4.4	7	74	18	11	255	4	0.6	0.12	-	-
1986	6.3	3	220	15	12	633	-	-	0.11	-	-
1987	3	9.5	334	15	4	143	-	0.25	0.25	0.1	94.1
1990	6.3	12	293	17	5	323	-	0.85	0.06	0.09	96.8
1991	7.8	10	749	19	10	701	-	0.7	0.09	0.09	93.7
1992	3.8	5	190	12	22	397	-	0.3	0.09	0.06	94.1
1993*	3	8	160	14	< 4	102	-	0.2	0.05	0.03	95

* 1993 results based on top 10 cm of core samples; all other years based on surface grab samples using Ekman dredge.

The attenuation of uranium and molybdenum in sediments with distance downstream are shown in Figures 1.15 and 1.16 and again shows the importance of the fen in reducing contaminant migration into the lower reaches of the Island Creek drainage. Swanson (1990) showed that 96% of the uranium and 85% of the molybdenum from the TMA does not reach the dolomite site on Island Creek downstream of the fen. Concentrations of radium-226, lead-210 and thorium in Island Lake sediments have not varied from baseline levels (Table 1.7).

Concentrations of arsenic, copper, molybdenum and uranium from sediment samples collected from Island Creek at Sandy Lake increased in 1993 compared to those measured in 1978. However, TAEM (1994) caution that these results are suspect because only one sediment sample in both 1978 and 1993 were collected at this site. Triplicate top 5 cm samples taken in 1995 by the SERM CEM program at the inlet to Sandy Lake and Sandy Lake just out from Island Creek (Island Creek Delta) likely confirm minor increases at Island Creek at Sandy Lake, for concentrations of arsenic, molybdenum and uranium (Appendix 8). These data may also indicate minor increases in concentrations of P, Ba, Co, Mg, Na, V, Zn, Pb-210, Po-210 and Ra-226, and larger increases in Ca, Fe and Mn. However, these data remain to be compared statistically between the CEM stations.

Figure 1.15: Concentrations of uranium in sediments at different locations within the Island Creek drainage during 1993 (after TAEM 1994).

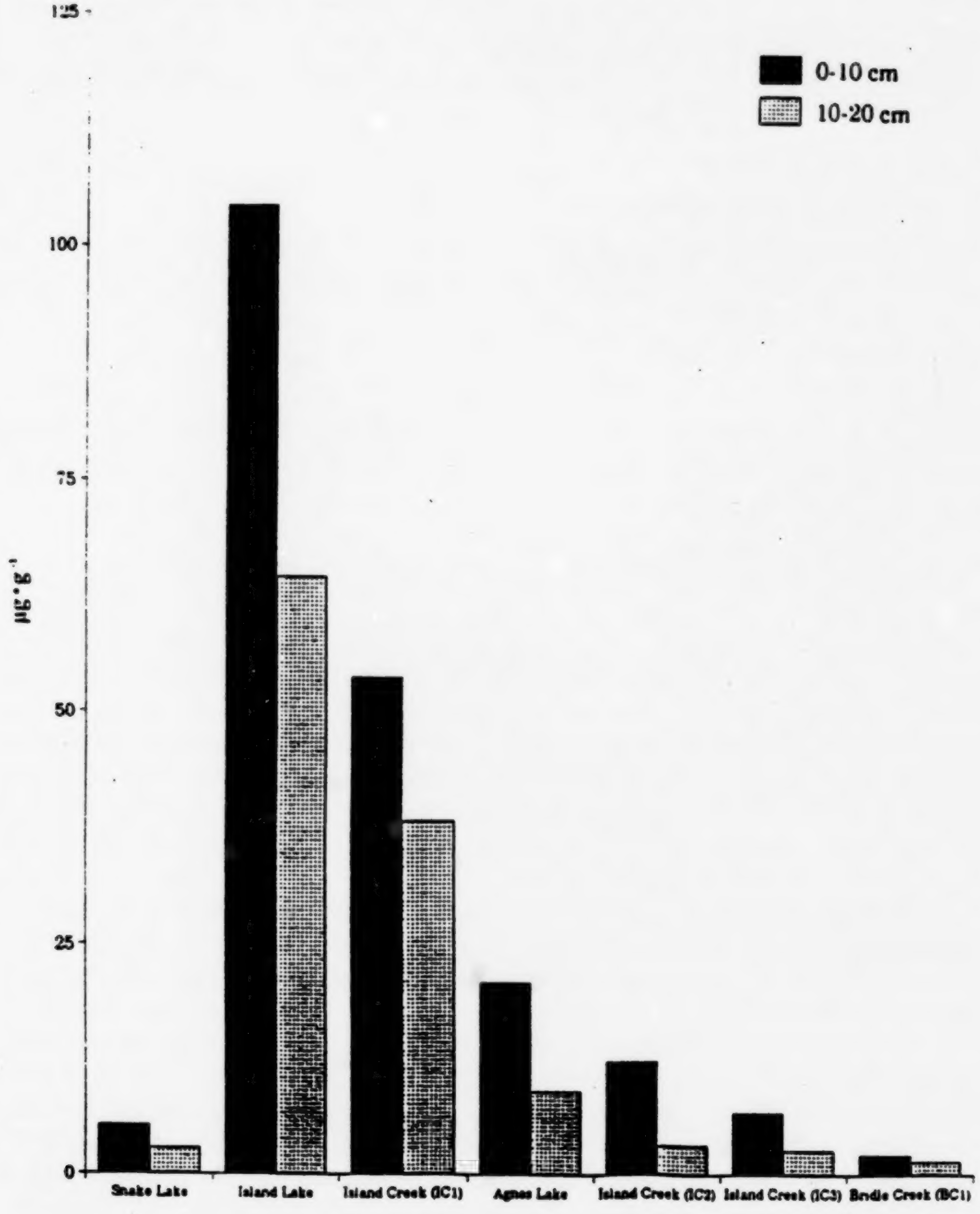
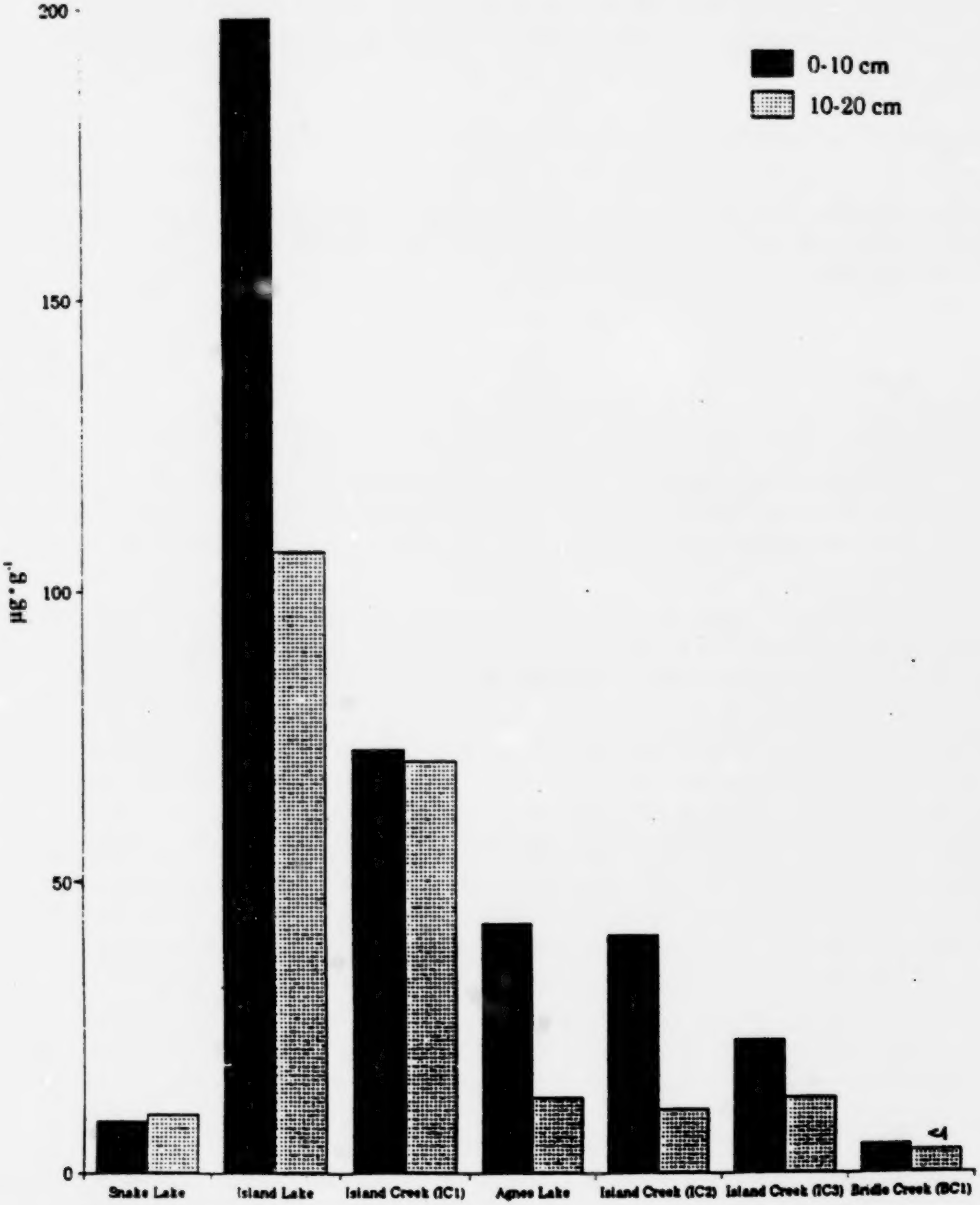


Figure 1.16: Concentrations of molybdenum in sediments at different locations within the Island Creek drainage during 1993 (after TAEM 1994).



In 1993 and 1994, sediments in Bridle Creek were sampled as a control, and in Agnes Lake to detect contamination caused by increased fen/Island Lake surface water levels directing new outflow to Agnes Lake. The new channel flows through a fen to Agnes Lake into a bay on the eastern side of the lake. At this point sediments show increased concentrations of molybdenum, lead and uranium when 1993 sediments were compared to 1994 sediments (Table 1.8).

Comparison of metal concentrations in 1994 Agnes Lake sediments to mean 1993-94 Bridle Creek control sediments (calculated from Table 1.8) shows that molybdenum is 29 times higher at 130 μg/L, lead is only 3 times higher at 28 μg/L, and uranium is 12 times higher at 30 μg/L. Concentrations of lead remain below the sediment quality guideline TEL of 35 μg/L (Smith *et al.* 1995).

1.3.2 Operations Areas

Boulder Creek sediments showed concentration peaks from baseline for uranium (12 up to 119 μg/g); radium-226 (0.16 up to 2.0 Bq/g) and lead-210 (0.12 up to 3.0 Bq/g) in 1982, during construction and development of the D ore body (Swanson, 1990). Since then, concentrations or uranium and radium-226 have declined to near baseline levels.

Peter River sediments were not analysed during the original baseline study. Increased concentrations of radium-226 (0.06 up to 0.25 Bq/g) in 1984 were attributed to construction and development of Phase II mines and roads in the vicinity (Swanson, 1990).

Sediments in Cluff Lake have shown some increases in concentrations of lead-210 and thorium-230 over baseline (Swanson, 1990). The 1988 survey also showed that there was a trend of higher concentrations of uranium, arsenic and lead at the north end of the lake compared to sites in the middle and south end. Those differences were attributed to mining activities (Swanson, 1990) and are also shown in Table 1.8 with the attenuation of contamination from Cluff Lake inlet site to Cluff Lake outlet site. In North Cluff Lake lead concentrations increased from a baseline concentration of 0.13-11.1 μg/g to 44 μg/g in 1993. Maximum concentrations of uranium, arsenic and lead were 42.3, 43.7 and 90 μg/g respectively in 1988. Concentrations of arsenic and lead have been above the guideline TEL's (5.9 and 35), and for arsenic have been above the probably effect level (PEL)of 17 μg/g (Appendix 7).

Table 1.8: Comparison of surface sediment data in the Cluff Lake area, 1988 to 1994

Location	Year	As	Co	Cu	Cr	Fe	Mn	Mo	Ni	Pb	Th	U	Zn	Pb-210	Ra-226	Th-230	LOI
		µg/g												Bq/g			%
Cluff Lake (site 1)	1988 (1)	43.7	20	39	52.3	176,333	767	< 4	34	90	11.3	42	150	0.7	0.28	-	39
Cluff Lake (site 1)	1993 (3)	25.0	22	56	34	55,000	2,400	19	31	44	-	38	130	0.5	0.35	0.3	34
Cluff Lake (site 2)	1988 (1)	38.0	21	29	40	177,667	713	< 4	34	64	9.8	20	143	0.73	0.4	-	31
Cluff Lake (site 2)	1993 (3)	9.7	24	41	32	55,000	5,700	23	40	57	-	25	150	0.75	0.35	0.25	33
Cluff Lake (site 3)	1988 (1)	25.7	18	51	48	186,000	1,007	< 4	36	63	13.7	19	180	1.3	0.3	-	41
Cluff Lake (site 3)	1993 (3)	20.0	20	54	31	55,000	550	16	32	44	-	19	150	0.85	0.25	0.2	38
Cluff Lake outlet	1990 (2)	2.5	-	5	-	-	-	4	16	13	-	6	59	0.55	0.13	0.01	15
Cluff Lake outlet	1991 (2)	1.2	-	5	-	-	-	4	7	7	-	1	29	0.148	0.04	0.052	10
Cluff Lake outlet	1992 (2)	4.1	-	22	-	-	-	4.6	25	15	-	4	61	0.118	0.056	0.076	25
Cluff Lake outlet	1993 (2)	4.3	-	20	-	-	-	4	18	4	-	4	90	0.236	0.102	0.076	32
First Lake (north)	1989 (1)	8.2	21	38	38	156,000	560	< 4	37	38	3.2	6	71	0.87	0.1	-	52
First Lake (south)	1989 (1)	4.6	19	37	37	10,467	223	< 4	26	3	3.7	8	83	0.37	0.11	-	41
First Lake (south)	1993 (3)	3.5	6	42	4	9,300	740	4	16	< 4	-	8	110	0.12	0.15	0.11	67
Fourth Lake	1989 (1)	7.6	14	36	36	56,333	220	< 4	17	14	6.5	25	72	1	0.28	-	36
Fourth Lake	1993 (3)	14.0	21	65	22	29,000	150	7	30	12	-	33	-	0.85	0.4	0.12	49
Snake Lake outlet	1990 (2)	0.4	-	1	-	-	-	4	2	7	-	1	1	0.13	0.02	0.03	1
Snake Lake outlet	1991 (2)	1.8	-	5	-	-	-	12	8	5	-	24	43	0.51	0.128	0.064	33
Snake Lake outlet	1992 (2)	2.1	-	7	-	-	-	9.4	8	27	-	17	95	0.3	0.102	0.086	32
Snake Lake outlet	1993 (2)	2.2	-	9	-	-	-	4	7	4	-	43	73	0.44	0.154	0.09	40
Snake Lake outlet	1993 (4)	2.1	-	9	-	-	-	11	16	< 4	-	8	-	0.14	0.13	0.05	-
Island Lake outlet	1990 (2)	6.3	-	12	-	-	-	293	17	5	-	323	37	0.85	0.06	0.09	68
Island Lake outlet	1991 (2)	7.8	-	10	-	-	-	748	19	10	-	701	44	0.7	0.092	0.066	69
Island Lake outlet	1992 (2)	3.8	-	5	-	-	-	190	12	22	-	397	51	0.3	0.086	0.06	71
Island Lake outlet	1993 (2)	7.7	-	10	-	-	-	354	20	4	-	284	53	0.59	0.116	0.07	64
Island Lake outlet	1993 (4)	3.1	-	9	-	-	-	73	15	< 4	-	54	-	0.19	0.1	0.02	-
Bridle Creek	1993 (4)	24.0	-	6	-	-	-	5	< 1	< 4	-	2	-	0.1	0.12	0.05	-
Bridle Creek	1994 (2)	3.4	-	< 1	-	-	-	< 4	5	15	-	3	37	0.12	0.06	0.06	37
Agnes Lake	1993 (4)	6.7	-	7	-	-	-	43	< 1	< 4	-	11	-	0.14	0.07	0.07	-
Agnes Lake	1994 (2)	5.3	-	< 1	-	-	-	130	4	28	-	30	56	0.2	0.03	0.04	36

(1) Swanson 1990, using average of top 10 cm from 3 replicate core samples

(2) Cluff Lake Annual Reports 1990 to 1994, using average of 5 replicate Ekman dredge samples

(3) TAEM 1993a, using 1 composite sample of 3 Ekman dredge grabs

(4) TAEM 1994, using average of top 10 cm from 3 composited core samples

In north Cluff Lake between 1988 and 1993 concentrations of uranium, arsenic, lead, and chromium have been decreasing, but manganese and molybdenum concentrations have shown increases from 767 to 2400 μg/g and <4 to 19 μg/g respectively. Iron concentrations have shown a 3.5 fold decrease from 176,333 to 55,000 μg/g (TAEM and SENES 1995).

1.3.3 Summary and Recommendations

1. Baseline collections and operational collections have been made using a number of methods and designs (eg. Ekman vs corers; stratified vs unstratified depth samples). Regardless of techniques, comparisons of surficial strata have been sufficient to detect general trends. It is recommended that sediment sampling be continued using techniques which will minimize loss of organic surface layers (eg. scuba or non impact coring) and concentrate on characterizing surficial sediments in which biological activity are centred. Replication needs to be sufficient to meet the requirements of appropriate statistical tests.

2. Poor return to site accuracy has been cited as a factor which could cause high error (eg. high year to year differences in organics content influencing metals content) (TAEM and SENES 1995). Methods increasing return to site accuracy are needed for improvement of monitoring program reliability.

3. Island Lake sediments have significantly increased concentrations of uranium and molybdenum but no sediment quality guidelines are available on which to base assessments or management decisions.

4. Since changes in sediment quality have occurred in the north end of Cluff Lake which have been attributed to mining activity, this portion of the lake should be monitored to assess trends in sediment contamination, biological effects and potential toxicity. However, in conducting monitoring in Cluff Lake, consideration should be given to the potential for changes in sediment quality due to inputs from natural sources, such as springs at the bottom of Cluff Lake, and along Peter Creek.

5. TAEM and SENES (1995) concluded that for the First to Fourth Lakes south of Cluff Lake, there are no indications of mining related effects. The lack of connectivity of these lakes to mining activities indicates that the requirement for continued monitoring should be reassessed.

6. Preliminary examination of data from a station in Sandy Lake suggest detectable but minor increases in sediment concentrations of arsenic, molybdenum, uranium, lead-210, pollonium-210, radium-226, phosphate, barium, cobalt, magnesium, sodium, vanadium

and zinc, and larger increases in calcium, iron and manganese.

7. In general, for the Cluff Lake area there are only minor accumulations of radionuclides in sediments, other than uranium in Island Lake.

1.4 Results of Biological Monitoring

1.4.1 Phytoplankton

Phytoplankton data have been collected on three separate occasions. Tones (1979) described baseline phytoplankton communities during 1978 and 1979. As part of that study, phytoplankton communities were sampled between June and October of 1978, and between June and May of 1979 in Cluff Lake, Sandy Lake, Island Lake and Snake Lake. The second survey was by Swanson (1990) who described phytoplankton communities during August 1988 (Cluff Lake), and May and July of 1989 (First to Fourth Lakes). Finally, TAEM (1994) collected phytoplankton from Cluff Lake and First to Fourth Lakes in September 1992. All of these studies collected phytoplankton using weighted Tygon tubing.

There have been no conclusions regarding impacts on phytoplankton.

1.4.2 Zooplankton

Melville (1994) examined the changes in zooplankton between baseline collections done in 1978 and 1979 and the community in 1989 in Island Lake. Community composition changed and was hypothesized to be related to salinity or to a lesser extent eutrophication. Calanoid copepoda disappeared, while cyclopoid copepoda and cladocerean species numbers have increased. Simpson's diversity index was much lower in 1989 (about 1.25) versus 1978-79 (about 2.75). Notably, *Ceriodaphnia reticulata,* a standard bioassay species, was absent in 1978-1979 but present in 1989 in contaminated conditions, and its high abundance was responsible for lowered diversity.

Baseline biomass of zooplankton was 704 μg/L compared to 99 μg/L in 1989. Melville (1994) suggested that changes in zooplankton, especially species composition, indicated that the lake is under stress. Changes in salinity at levels that can cause acute toxicity and restrict reproduction of calanoid copepoda were suggested as the main factor for reduced diversity. Increases in concentrations of ammonia, uranium, molybdenum and manganese may also have been party responsible for ecological changes.

No other zooplankton studies have been conducted.

1.4.3 Macrophyte Community Structure and Density

Baseline studies for macrophytes were completed in 1980 for Snake and Island Lake (Abouguendia *et al.* 1980), and post-operational mapping of macrophytes were done in Cluff Lake in 1988-89 (Swanson 1990). Changes in macrophytes were reassessed by Hynes (1990) comparing 1980 to 1986 data. TAEM (1994), using data collected in 1993, reassessed changes in community structure. Each survey consisted of:

- linear lake transects
- shoreline transects
- stem counts

There are 22 linear lake transects established in Island Lake which have been permanently marked. These transects are used to map and record the macrophyte community from shore to 35 m into the lake at 1-m intervals. Depth and distance from shore are recorded. Six shoreline transects are also permanently marked at which both shoreline and aquatic plant species numbers and cover were recorded for 15 quadrants. Finally, at one of the permanent stations, stem counts of various species have been conducted.

Analysing the three years of data, TAEM (1994) used different statistical tests to compare macrophyte communities among years and stations. For the linear lake transect data, they compared presence/absence of each species using contingency table analysis. That analysis would effectively determine if the presence/absence of a given species had changed from year to year. Using the shoreline transect data, TAEM (1994) used a two-way analysis of variance (ANOVA) to test whether there had been changes over time. For the species specific stem counts, TAEM (1994) used ANOVA. This analysis is adequate for single-species counts.

Hynes (1990) found changes in Island Lake consistent with shifts caused by salinity with eleven species either decreasing in abundance or eliminated, and two salt tolerant species increasing in abundance. Nine species showed no change. TAEM (1994) found that based on lake linear transects 11 species were eliminated and 5 new species were present, relative to 1980. TAEM (1994) also found that four species not previously described in shoreline transects in 1980 were present in 1993, and that four species were eliminated. Stem density of the viscid great bulrush, *Scirpus acutus* had significantly increased in comparison to 1980 and 1986 in 1993.

TAEM (1994) also attributed changes in Island Lake to increased salinity by identifying reductions in populations of salt intolerant species such as ribbon leaved pond weed (*Potamogeton epihydrus*) and arum-leaved arrowhead (*Sagittaria cuneata*) and increases of salt tolerant species such as stonewart (*Chara sp.*), Richardson's pondweed (*P. richardsoni*), viscid great bullrush, and mare's tail (*H. vulgaris*).

Permanent riparian aquatic sample plots have been set up at three locations in the fen to determine long term changes in species abundance and density (TAEM 1994). One station is located on Island Creek close to the Agnes Lake outlet.

The distribution of aquatic macrophyte species was mapped in the Cluff Lake drainage in 1988-89 (Swanson 1990) and detailed species lists were compiled in the Cluff drainage in 1992-93 (TAEM 1993), and in the Island Creek/Cluff Lake combined drainages in 1993 by Godwin and Ternier (1993). However, no assessment of changes to the macrophyte community structure has been done for the Cluff Lake drainage.

1.4.4 Benthos Community Structure and Density

Benthic communities have been surveyed on four occasions (1978/79, 1985, 1988/89, 1992/93). Both lake and creek habitats have been surveyed, and a variety of sampling methods have been employed for both. In general, surveys have employed at least some common methodologies which allows for comparisons among years. However, there have also been some differences which TAEM (1994) pointed out could lead to difficulty in direct comparison. Lake habitats have been surveyed using Ekman grabs and sweep nets (Hynes, 1990) although Ekman samples were common to all years. TAEM (1994) did not use sweep nets to sample lake habitats in 1993. Stream samples have been collected using Neill Cylinders (1993), sweep nets (1985), Hester-Dendy multiplate samplers (1985), drift nets (1978/79, 1985, 1988/89) and Surber samplers (1978/79, 1988/89). During Hynes' study, sampling of the creeks involved the use of drift nets and Hester-Dendy multiplate samplers only. Neither of these was used during the baseline studies or the 1993 survey. Mesh size, of sieves used to wash Ekman grab samples (which can affect estimated densities of organisms) was larger in 1978/79 (752 μm) than in 1985 or 1993 (600 μm). It is uncertain whether this difference could explain some of the observed differences in community composition between years. It is also uncertain whether a similar difference exists for the stream samples collected using sweep nets, Surber samplers or Neill cylinders. The Neill cylinders used during the 1993 survey had a mesh size of 210 μm.

Other inconsistencies between survey years that TAEM (1994) highlighted and inferred might result in data being less than optimal for comparison include:

- a change in taxonomists during each survey
- differences in locations sampled

Surveys conducted within the Island Lake system have been fairly conscientious about re-surveying the same locations. Repeat surveys have occurred within Island Lake, Snake Lake, Island Creek, and Boulder Creek.

There are few baseline data for the Cluff Lake drainage system. The only benthos from this drainage are samples from Boulder Creek (Tones, 1979; SRC 1982 data), Peter River (SRC 1982 data) and Cluff Creek at the outlet of Cluff Lake (Tones, 1979). No baseline samples were collected in Cluff Lake itself, or First, Second, Third or Fourth Lakes. There are also no data on Germaine Lake.

There is only one station on the mine site that is not exposed to runoff from the mine (Bridle Creek upstream of Agnes Lake). Samples from this station could be used as temporal controls for other creek stations being monitored. There are no temporal controls for any of the lake stations being monitored.

TAEM (1994) summarized the benthic data collected over the three years for the Island Lake system. In part to avoid inconsistencies between taxonomists, they chose to examine changes in major taxonomic groups. They also calculated some common indices of community structure:

- Shannon-Wiener diversity
- Equitability
- Margalef's Richness
- Simpson's Dominance

Although TAEM (1994) indicated that there were differences in substrate between stream locations, there was no attempt to determine if differences in substrate might have caused the observed between-station differences.

Tables 1.9 to 1.11 show the temporal changes in density and community structure of benthos collected between baseline (1974-75) to 1992-1993 for Island Lake, Snake Lake and Cluff Lake.

In Island Lake there is a major increase in density of metals/pollution tolerant chiromomidae (Table 1.9). In 1993 chiromomidae represented 97% of the organisms sampled versus only 19.4% in 1978. Species loss is also apparent for leeches, amphipods, fingernail clams, snails and possibly aquatic beetles. Reductions in the density of mayflies, cladocerans and aquatic earthworms are also evident. New taxa of copepoda and ostracoda were also apparent. (note: some data include zooplankton such as cladocera and copepoda, usually not considered to be benthos).

Since 1978, there has been a significant increase in the density of organisms (combined taxa from 1797.00/ m^2 to 4441.00/m^2). This increase is largely attributable to chiromomids.

Benthic communities like the one found in 1993 are typical of locations with severely degraded water quality. Both Swanson (1991) and Hynes (1990) suggested the changes in the benthic community of Island Lake were principally caused by increased salinity rather than by high concentrations of metals and radionuclides.

In Snake Lake (Table 1.10) the benthic community in 1985 was different than the community in 1978/79 and 1993. However, the nature of the difference in 1985 is difficult to interpret. Amphipods were the dominant member of the benthic community in 1985 with no clams, gastropods, hirudinea or oligochaetes. The reoccurrence of these groups in Snake Lake in 1993 is either remarkable, questions the accuracy of the 1985 data, or suggests that the 1985 samples were collected in a habitat that was different than in the other years. Groups such as clams and snails do not have life cycles which are optimal for colonization (in contrast to insects like chiromomids) and it is unlikely that number could have increased to the observed densities in 1993 unless there had been individuals present in 1985.

Table 1.9: Comparison of taxa identification and enumeration (organisms/m^2) of benthic macroinvertebrates collected, using an Ekman dredge, from Island Lake in 1978, 1985 and 1993

Taxon	**Number of organisms/m^2**			
	1978[1]	**1979[1]**	**1985[2]**	**1993[3]**
Phylum: Arthropoda	-	-	-	-
Class: Insecta	-	-	-	-
Order: Ephemeroptera (mayflies)	-	-	96.9	6.5
Order: Odonta (dragonflies)	-	-	10.8	2.2
Order: Coleoptera (beetles)	-	-	7.2	-
Order: Trichoptera (caddisflies)	2.1	-	3.6	26.9
Order: Diptera (true flies)	348.8	701.0	1366.8	4310.8
Subphylum: Chelicerata	-	-	-	-
Class: Arachnida (spiders and mites)	-	-	-	-
Order: Hydracarina (water mites)	4.3	12.0	-	3.2
Subphylum: Crustacea	-	-	-	-
Class: Brachiopoda	-	-	-	-
Order: Cladocera (water fleas)	-	184.2	-	2.2
Class: Malacostraca	-	-	-	-
Order: Amphipoda (sideswimmers)	1320.6	287.1	93.3	-
Subclass: Copepoda (copepods)	-	-	-	43.0
Class: Ostracoda (seed shrimps)	-	-	-	5.4
Phylum: Mollusca	-	-	-	-
Class: Bivalvia (clams)	25.5	232.1	57.4	-
Class: Gastropoda (snails)	2.1	2.4	-	-
Phylum: Annelida (segmented worms)	-	-	-	-
Class: Hirudinea (leeches)	8.5	12.0	-	-
Class: Oligochaeta (aquatic earthworms)	80.8	129.2	-	8.6
Phylum: Nematoda (roundworms)	4.3	7.2	-	32.2
Total number of organisms/m^2	**1797.0**	**1567.0**	**1635.9**	**4441.0**
Number of taxa[4]	**25.0**	**31.0**	**22.0**	**31.0**
Shannon-Wiener Diversity ($\log_{10}$)[5]	**0.6**	**1.1**	**0.7**	**0.7**

[1] Tones 1979 in Hynes 1985

[2] Hynes 1985

[3] TAEM 1994

[4] calculated from detailed Table 3.6 in TAEM 1994

[5] taken from Table 3.8, TAEM 1994

Table 1.10: Comparison of taxa identification and enumeration (organisms/m^2) of benthic macroinvertebrates collected, using an Ekman dredge, from Snake Lake in 1978, 1985 and 1993

Taxon	**Number of organisms/m^2**			
	1978[1]	**1979**[1]	**1985**[2]	**1993**[3]
Phylum: Arthropoda	-	-	-	-
Class: Insecta	-	-	-	-
Order: Odonta (dragonflies)	-	2.4	-	-
Order: Trichoptera (caddisflies)	7.2	4.8	5.4	8.1
Order: Lepidoptera (butterflies and moths)	-	-	-	1.6
Order: Diptera (true flies)	519.2	311.0	166.8	190.4
Order: Hemiptera (true bugs)	-	2.4	-	-
Subphylum: Chelicerata	-	-	-	-
Class: Arachnida (spiders and mites)	-	-	-	-
Order: Hydracarina (water mites)	2.4	-	-	-
Subphylum: Crustacea	-	-	-	-
Class: Brachiopoda	-	-	-	-
Order: Cladocera (water fleas)	-	9.6	-	-
Class: Malacostraca	-	-	-	-
Order: Amphipoda (sideswimmers)	382.8	318.2	791.0	129.1
Subclass: Copepoda (copepods)	-	260.8	-	1.6
Class: Ostracoda (seed shrimps)	-	-	-	8.1
Phylum: Mollusca	-	-	-	-
Class: Bivalvia (clams)	545.5	507.1	-	263.0
Class: Gastropoda (snails)	16.8	28.7	-	250.1
Phylum: Annelida (segmented worms)	-	-	-	-
Class: Hirudinea (leeches)	14.4	7.2	-	1.6
Class: Oligochaeta (aquatic earthworms)	19.1	47.9	-	24.2
Total number of organisms/m^2	**1507.2**	**1500.0**	**963.2**	**877.7**
Number of taxa[4]	**20.0**	**23.0**	**12.0**	**27.0**
Shannon-Wiener Diversity ($\log_{10}$)[5]	**0.7**	**0.7**	**0.4**	**0.9**

[1] Tones 1979 in Hynes 1985

[2] Hynes 1985

[3] TAEM 1994

[4] calculated from detailed Table 3.6 in TAEM 1994

[5] taken from Table 3.8, TAEM 1994

Table 1.11: Comparison of taxa identification and enumeration (organisms/m^2) of benthic macroinvertebrates collected, using an Ekman dredge, from Cluff Lake in 1974-1975, 1988 and 1992

Taxon	Number of organisms/m^2								
Station (1)	D5	1A	1	D41b	2A	2	D55b	3A	3
Year	1974-75 (2)	1988 (3)	1992 (4)	1974-75 (2)	1988 (3)	1992 (4)	1974-75 (2)	1988 (3)	1992 (4)
Phylum: Arthropoda	-	-	-	-	-	-	-	-	-
Class: Insecta	-	-	-	-	-	-	-	-	-
Order: Odonata (dragonflies)	20.0	-	-	-	-	-	-	-	-
Order: Trichoptera (caddisflies)	-	-	6.4	-	-	-	-	-	-
Order: Plecoptera (stoneflies)	20.0	-	-	-	-	-	-	-	-
Order: Coleoptera (beetles)	-	-	6.4	-	-	-	-	-	-
Order: Ephemeroptera (mayflies)	280.0	-	6.4	-	-	-	-	-	-
Order: Diptera (true flies)	100.0	357.3	359.0	240.0	350.9	269.2	780.0	580.5	262.8
Subphylum: Chelicerata	-	-	-	-	-	-	-	-	-
Class: Arachnida (spiders and mites)	-	-	-	-	-	-	-	-	25.6
Subphylum: Crustacea	-	-	-	-	-	-	-	-	-
Class: Brachiopoda	-	-	-	-	-	-	-	-	-
Order: Cladocera (water fleas)	-	-	-	-	-	-	-	-	6.4
Class: Malacostraca	-	-	-	-	-	-	-	-	-
Order: Amphipoda (sideswimmers)	60.0	-	-	-	-	-	-	-	6.4
Subclass: Copepoda (copepods)	-	-	6.4	-	-	25.6	-	-	12.8
Class: Ostracoda (seed shrimps)	-	-	19.2	-	-	-	-	-	32.1
Phylum: Mollusca	-	-	-	-	-	-	-	-	-
Class: Gastropoda (snails)	-	299.8	-	-	-	-	60.0	-	38.5
Class: Bivalvia (clams)	60.0	-	724.4	560.0	523.1	282.1	280.0	312.6	339.7
Phylum: Annelida (segmented worms)	-	-	-	-	-	-	-	-	-
Class: Oligochaeta (aquatic earthworms)	-	12.8	19.2	40.0	12.8	44.9	-	31.9	-
Phylum: Nematoda (roundworms)	-	-	-	-	-	-	-	-	6.4
Phylum: Nematomorpha (horsehair worms)	-	6.4	-	-	6.4	-	-	-	-
Total number of organisms/m^2	**540.0**	**676.2**	**1147.4**	**840.0**	**893.1**	**621.8**	**1120.0**	**925.0**	**730.8**
Number of taxa[4]	**6**	**4**	**8**	**3**	**4**	**4**	**3**	**3**	**9**
Shannon-Wiener Diversity ($\log_{10}$)	**0.6**	**0.4**	**0.4**	**0.3**	**0.3**	**0.5**	**0.3**	**0.3**	**0.6**
Margalef's Richness (LN)	**0.8**	**0.5**	**1.0**	**0.3**	**0.4**	**0.5**	**0.3**	**0.3**	**1.2**

(1) Sampling station designations correspond to original documents. Each temporal set was collected at approximately the same location

(2) Sawchyn 1975

(3) Swanson 1990

(4) TAEM 1992

However, the similarity between the community composition and diversity found in 1993 and baseline, suggests that impacts in Snake Lake are not severe. In 1993, gastropods did comprise a larger portion of the community than during baseline. Since snails graze on periphyton, their increased numbers suggest an increase in the periphytic food supply.

Cluff Lake benthos data in Table 1.11 summarize data collected since baseline in 1974-75. Swanson (1991) discussed the 1988-89 study on Cluff Lake and concluded that diversity has not appeared to change from baseline but cautioned that the baseline for Cluff was very limited, and the taxonomy was open to question. TAEM 1993 data were collected at the north, middle and south areas of Cluff Lake and show a slightly richer more diverse assemblage at the south end of Cluff Lake. The use of appropriate multivariate statistical techniques and examination of species dominance/pollution tolerance relationships would be necessary in order to determine the ecological significance of changes. Monitoring should be continued to determine changes in benthos in Cluff Lake.

Stream data are much less comparable. TAEM (1994) suggests this was because of a lack of similarity of methods. TAEM (1994) therefore chose to discuss the occurrences in the system of taxa that are known to be sensitive to disturbances. They suggested that the presence of *Baetis* (Ephemeroptera) and stoneflies (Plecoptera) in Island Creek during 1985 and 1993 provided evidence that the Island Creek community has not changed appreciably. In contrast, the Snake Creek community did not have the same complement of sensitive species suggesting it has been impacted by treated effluent. TAEM and SENES (1995) recommended establishment of a well designed benthos monitoring program in the Island Lake system. Particular attention should be paid to detecting changes in Agnes Lake and Island Creek below Island Lake.

Swanson's (1991) summary of the 1988-89 survey suggested that there have been changes in the benthic fauna of Boulder Creek over time particularly at a site downstream towards Cluff Lake. Swanson (1991) concluded that the benthic community may have changed as a result of physical habitat disturbance. However, the results may have been caused by difference in sampling seasons among the years surveyed.

Swanson (1991) also suggested that changes in the invertebrate fauna in the Peter River, Earl Creek and Claude Creek were caused by human activity. The operation of a cement batch plant and exploration were proposed as having caused changes in the Earl Creek benthic community. During exploration of the DP ore body, wastewater containing drilling mud and oily substances entered the creek (only once) despite the presence of sand dykes. According to Swanson (1991), a depauperate benthic community persisted all the way to Fletts Pond above Cluff Lake. The changes in the Peter River were difficult to attribute to any specific disturbance (Swanson 1991). In Claude Creek, high numbers of fingernail clams were present during a 1982 survey. During the 1989 survey, they were reduced in numbers. Swanson

(1991) was also unable to specify the cause of this change.

1.4.5 Fish Community Structure, Density and Biomass

Several fish community surveys have been conducted to date. Baseline data were collected by Tones (1979) in 1978 for Snake, Island and Sandy Lakes. Swanson (1990) collected fish from the Cluff Lake drainage in 1988-89. TAEM (1993) collected spawning fish in the fall of 1992 and the spring of 1993 within the Cluff Lake drainage, while TAEM (1994) collected larger species from Snake, Island and Agnes Lakes during 1993. The SERM CEM program sampled Sandy Lake and a control lake 32 km above Sandy Lake on the Douglas River in 1995 to assess status of the Sandy Lake fishery. Seining in the control lake and in Sandy Lake was conducted for comparison of small size classes and smaller species. Data on length-weight relationships for fish from Third and Fourth Lakes are presented in Amok (1992). Swanson (1990) collected fish using seines and gill nets and therefore described both the small and large species in the Cluff Lake drainage. The two surveys by TAEM (1993, 1994) used only gill nets and collected only the larger species. Most of the surveys recorded length, weight, age, sex and stomach contents of collected fish.

No statistical analyses have been done to summarize the variations of fish community composition within the Cluff Lake and Island Lake drainages. Swanson (1990) analysed length-weight relationships and growth curves for pike (*Esox lucius*), cisco (*Coregonus artidii*) lake trout (*Salvelinus namaycush*) and walleye (*Stizostedion vitreum*), and summarized changes over time and differences among lakes within the Cluff Lake drainage.

Swanson (1990) suggested that within Cluff Lake, there had been no changes in fish species composition since 1974-75. In the Island Lake system it is difficult to determine if there had been changes in the fish community because of differences in fishing effort (baseline surveys used a combination of beach seining and gill netting while the 1993 survey used only gill netting) (TAEM, 1994). In addition, no comparisons of growth rates or condition factors were made, so that it is impossible to determine if there were sublethal effects on populations of large fish species (TAEM, 1994). TAEM suggested that there were too few observations collected during baseline surveys for comparison against the 1993 survey.

Snake Lake catch results (TAEM and SENES 1995) were comparable to other lakes near the study area (Table 1.12), and to total numbers and weight and fish caught in Sandy Lake in 1995 (Table 1.13). However, TAEM and SENES (1995) note a sixfold increase in biomass between 1978 baseline and 1993 for Snake Lake data, attributed to a marked increase in northern pike captured. Northern pike were dominant in 1978 (95%) and in 1993 (83%). The only other species caught was a single white sucker and no burbot were caught even though these were captured in Snake Lake by Abouquendia *et al.* 1979.

Table 1.12: Comparison of the average number and weight of fish captured per standard gang gillnet of selected waterbodies in the Cluff Lake area

Water body	Year		Average number of fish caught per net	Catch weight (kg)
Agnes Lake	1993	(1)	32.0	27.4
Cluff Lake	1974-1975	(2)	10.6	6.9
Cluff Lake	1988	(3)	11.9	10.8
Germaine Lake	1978	(4)	10.0	7.0
Island Lake	1978	(4)	36.5	42.1
Island Lake	1993	(1)	132.0	116.0
First Lake	1989	(3)	21.0	17.0
Second Lake	1989	(3)	12.5	15.3
Third Lake	1982	(3)	60.0	28.2
Third Lake	1989	(3)	18.0	17.8
Fourth Lake	1989	(3)	12.0	12.1
Sandy Lake	1978	(4)	40.5	40.9
Snake Lake	1978	(4)	15.1	4.8
Snake Lake	1993	(1)	12.0	30.7
Mean of lakes			30.3	26.9

(1) TAEM 1994
(2) Sawchyn 1975
(3) Swanson 1990
(4) Tones 1979

Table 1.13: Numbers and total weights (kg) of fish caught in three half-standard gang gillnets in Sandy Lake, 1995. Comparison with 1979 baseline study results.

	Northern Pike		Walleye		Lake Whitefish		White Sucker		Total	
	No.	Wt.	No.	Wt.	No.	Wt.	No.	Wt.	No.	Wt.
SYSG1	8	6.8	13	11.1	26	18.4	2	1.9	49	38.2
SYSG2	3	3.1	8	6.2	2	0.2	0	0	13	9.5
SYSG3	2	0.7	0	0	2	2.4	1	1.7	5	4.2
Total	13	10.5	21	17.2	30	21.0	3	3.6	66	51.8
average/set	4.3	3.5	7.0	5.7	10.0	7.0	1.0	1.2	22.0	17.2
1979 average	2.4	4.3	4.8	4.7	10.1	9.6	1.3	1.2	20.1	20.4
% of total	19.7%	20.3%	31.8%	33.3%	45.5%	40.6%	4.5%	6.9%	100%	100%
1979 % of total	11.8%	20.8%	23.6%	22.9%	50.3%	47.2%	6.2%	8.7%	100%	100%

Unpublished data from Saskatchewan Environment and Resource Management, Cumulative Effects Monitoring Program, 1995

In Island Lake TAEM and SENES (1995) showed that a major impact on population structure was evident and production increased dramatically. The total number of fish increased 3.6 fold and total weight increased 2.75 fold between 1978 and 1993 (Table 1.12). Representation of northern pike decreased to 3% and 13% in number and biomass of total catch, respectively, versus 86 % and 91% in 1978, while white sucker increased to 97% and 89% of the total catch by number and weight, respectively, versus baseline of 14% and 9%, respectively. Major shifts of this nature are indicative of reduced predation pressure on the white sucker population from northern pike which are intolerant of conditions in the lake. Comparison of catch and species representation between 1978 and 1995 for Sandy Lake do not show any major shift in large fish species (Table 1.13).

1.4.6 Bioaccumulation of Metals and Radionuclides

1.4.6.1 Macrophytes

Concentrations of metals (Al, As, Cu, Mo, Ni, Pb, U, Zn) and radionuclides (Th-230, Pb-210, Ra-226) have been determined in yellow pond lilies (*Nuphar variegatum*) collected from Island Lake and Snake Lake (1980-86) and beaked sedge (*Carex rostrata*) collected from Island Creek at Sandy Lake and Boulder Creek at Cluff Lake (1980-85) (Hynes, 1990). In 1988, Amok (1992) collected sedges from Boulder Creek and Cluff Lake and pond lilies from Cluff Lake. In 1992, TAEM (1993) collected macrophytes from Snake Lake, Island Lake, the fen west of Island Lake, Agnes Lake, and Unnamed Lakes 1 and 2 (i.e. small lakes on Island Creek below Island Lake).

Hynes (1990) reported that uranium was the contaminant that increased the most in concentration over time and that concentrations were highest in yellow pond lilies from Island Lake. Elevated concentrations of uranium were also found in sedges collected from Boulder Creek. High levels of uranium in sedges from Boulder Creek were coincident with mining of the D ore body. During mining of the D ore body, radium-226 concentrations were also high in sedges from Boulder Creek but not in Cluff Lake.

The 1988 survey suggested that uranium and thorium concentrations in sedges from Boulder Creek were still higher than baseline levels, but were lower than the peak concentrations observed in 1983-84. Concentrations of all contaminants in lilies from Cluff Lake were lower than concentrations in either lilies or sedges from Boulder Creek in 1988.

TAEM (1994) reported that uranium and molybdenum concentrations were higher in plants collected in Island Lake and the fen west of Island Lake compared to samples obtained from other sites. Arsenic and radium-226 concentrations were also higher in Island Lake macrophytes. Uranium concentrations were generally higher than background concentrations. Observed radionuclide concentrations were not as high in 1993 as had been observed in previous years. It was suggested that this may be due to slightly different collection methodologies in 1993 (plants clipped above the roots) compared to previous years (whole plants, including roots, collected).

1.4.6.2 Fish

Tissue concentrations of metals (Al, As, Cu, Mo, Ni, Pb, U, Zn) and radionuclides (Pb-210, Po-210, Ra-226, Th-230) have been determined for fish from Cluff Lake (northern pike, lake whitefish), Island Lake (northern pike), Sandy Lake (northern pike, lake whitefish) between 1985 and 1993. Background (pre-1984) tissue concentration data exist only for uranium, Pb-210 and Th-230. Generally, flesh and bone from 4 to 5 fish have been subjected to analysis. A change in analytical procedures to avoid contamination during ashing was made as of 1988 for trace elements. Comparison of pre-and post 1988 data meant having to convert post 1988 data to ash weight basis.

Swanson (1991) assessed trace element concentrations in fish from all three lakes between 1980 to 1989 and showed an apparent increase. The only radionuclide to have increased in concentration was Pb-210 in Sandy Lake white fish flesh. These increases could not be unequivocally attributed to bioaccumulation since they may have been artifacts of analytical changes.

TAEM and SENES (1995), using factorial ANOVA, examined the effect of site and year (1989, 1991, 1993) on tissue concentrations. Analyses supported the hypothesis of Swanson (1991) that metal concentration increases were artifacts of changes in laboratory procedures since a consistent decrease in concentrations of most parameters is clearly evident. For trace metals, only molybdenum is bioaccumulative in Island Lake in northern pike (Table 1.14). Of the radionuclides only uranium is bioaccumulative in northern pike flesh and bone from Island Lake (Table 1.15). There was also evidence of uranium bioaccummulation in lake whitefish flesh and bone but sample size was too small for unequivocal results.

1.4.7 Summary and Recommendations

1. Phytoplankton monitoring has been conducted but no conclusions have been made regarding impacts.

2. Zooplankton in Island Lake have shown major changes in species composition, decreased diversity and order of magnitude reduction in biomass between 1978/79 to 1989. These differences indicated that the lake was under stress and Melville (1994) hypothesized that salinity could be the main cause. Increased concentrations of ammonia, uranium, molybdenum and manganese may also have been partly responsible for the impacts.

Table 1.14: Trace metal chemical analyses of northern pike bone and flesh from three water bodies in the Cluff Lake area, 1989, 1991 and 1993. Results reported on a wet weight basis.

Parameter µg/g	Year	Bone									Flesh								
		Cluff Lake			Island Lake			Sandy Lake			Cluff Lake			Island Lake			Sandy Lake		
		Mean	Std.	N	Mean	Std.	N	Mean	Std.	N	Mean	Std.	N	Mean	Std.	N	Mean	Std.	N
Al	1989	7.9	5.5	5	13.1	10.8	5	4.5	5.9	5	4.4	4.1	5	1.9	1.2	5	1.1	0.7	5
	1991	2.4	0.9	5	6.5	-	1	3.2	0.7	5	0.5	0	5	0.5	-	1	0.5	0	5
	1993	2.2	0.2	4	3.8	2.2	4	4.0	1.2	5	1.3	0.6	4	2.2	1.0	4	1.9	0.5	5
As	1989	0.08	0.06	5	0.05	0.02	5	0.06	0.02	5	0.05	0.02	5	0.03	0.01	5	0.13	0.16	5
	1991	0.05	0	5	0.05	-	1	0.05	0	5	0.02	0	5	0.02	-	1	0.02	0	5
	1993	0.05	0.002	4	0.06	0.01	4	0.06	0.003	5	0.02	0.005	4	0.07	0.05	4	0.05	0.02	5
Cu	1989	0.28	0.07	5	0.34	0.26	5	0.31	0.11	5	0.20	0.05	5	0.44	0.23	5	1.12	0.48	5
	1991	0.30	0.10	5	0.20	-	1	0.20	0	5	0.14	0.09	5	0.20	-	1	0.12	0.04	5
	1993	0.23	0.05	4	0.20	0.01	4	0.26	0.05	5	0.10	0.00005	4	0.20	0.0003	4	0.18	0.04	5
Mo	1989	0.88	0.14	5	0.96	0.29	5	0.98	0.34	5	0.49	0.07	5	0.58	0.14	5	0.44	0.09	5
	1991	0.10	0	5	0.30	-	1	0.10	0	5	0.02	0.004	5	0.02	-	1	0.02	0	5
	1993	0.10	0.004	4	0.40	0.22	4	0.10	0.01	5	0.02	0.0002	4	0.02	0.01	4	0.02	0.0005	5
Ni	1989	0.89	0.83	5	1.31	1.06	5	0.52	0.62	5	0.90	0.58	5	0.78	0.48	5	0.85	0.58	5
	1991	0.05	0	5	0.05	-	1	0.05	0	5	0.02	0.004	5	0.02	-	1	0.02	0	5
	1993	0.05	0.002	4	0.05	0.004	4	0.05	0.01	5	0.03	0.02	4	0.02	0.0003	4	0.03	0.01	5
Pb	1989	0.97	0.24	5	0.87	0.15	5	0.87	0.12	5	0.54	0.14	5	0.58	0.14	5	0.44	0.09	5
	1991	0.12	0.04	5	0.10	-	1	0.10	0	5	0.06	0.02	5	0.05	-	1	0.05	0	5
	1993	0.10	0.004	4	0.10	0.004	4	0.10	0.01	5	0.05	0.0003	4	0.05	0.0004	4	0.05	0.0003	5
U	1989	0.042	0.014	5	2.685	0.827	5	0.045	0.013	5	0.004	0.001	5	0.065	0.023	5	0.006	0.002	5
	1991	0.026	0.013	5	2.200	-	1	0.064	0.098	5	0.002	0	5	0.054	-	1	0.006	0.004	5
	1993	0.020	0.020	4	3.223	1.703	4	0.018	0.013	5	0.002	0.001	4	0.059	0.027	4	0.002	0.001	5
Zn	1989	76.2	11.2	5	52.6	15.3	5	74.4	10.4	5	19.6	9.5	5	15.4	7.2	5	15.4	11.7	5
	1991	32.0	12.7	5	24.0	-	1	26.8	3.6	5	5.6	1.1	5	3.4	-	1	4.5	2.0	5
	1993	52.0	7.4	4	37.5	3.4	4	53.6	10.3	5	4.8	0.6	4	4.5	0.4	4	4.8	0.3	5

Table 1.15: Radionucleide chemical analyses of northern pike bone and flesh from three water bodies in the Cluff Lake area, 1989, 1991 and 1993. Results reported on a wet weight basis.

Parameter Bq/g	Year	Bone									Flesh								
		Cluff Lake			Island Lake			Sandy Lake			Cluff Lake			Island Lake			Sandy Lake		
		Mean	Std.	N	Mean	Std.	N	Mean	Std.	N	Mean	Std.	N	Mean	Std.	N	Mean	Std.	N
Pb-210	1989	0.002	0	5	0.002	0	5	0.002	0	5	0.002	0	5	0.002	0	5	0.002	0	5
	1991	0.002	0	5	0.002	-	1	0.002	0	5	0.001	0	5	0.001	-	1	0.001	0	5
	1993	0.002	0.00001	4	0.002	0.001	4	0.002	0	5	0.001	0	4	0.001	0	4	0.001	0	5
Po-210	1989	0.0038	0.0015	5	0.0017	0.001	5	0.0009	0	5	0.0055	0.0043	5	0.0026	0.0019	5	0.0024	0.0011	5
	1991	0.0005	0	5	0.0005	-	1	0.0005	0	5	0.0003	0	5	0.0003	-	1	0.0003	0	5
	1993	0.0025	0.0023	4	0.0014	0.0006	4	0.0013	0	5	0.0026	0.0013	4	0.0014	0.0008	4	0.0019	0.0012	5
Ra-226	1989	0.0006	0.0001	5	0.0024	0.0010	5	0.0021	0.0010	5	0.0001	0.00001	5	0.0001	0.00003	5	0.0001	0.0001	5
	1991	0.0014	0.0005	5	0.003	-	1	0.0026	0.0019	5	0.0001	0.00002	5	0.0001	-	1	0.0001	0.0001	5
	1993	0.0018	0.0010	4	0.004	0.0016	4	0.0024	0.0011	5	0.00005	0.00001	4	0.0003	0.0004	4	0.0001	0.0001	5
Th-230	1989	0.0012	0.0002	5	0.002	0.0001	5	0.0019	0.0006	5	0.0002	0.0001	5	0.0001	0.00001	5	0.0001	0.00002	5
	1991	0.0026	0.0009	5	0.002	-	1	0.0024	0.0005	5	0.0002	0.0002	5	0.0001	-	1	0.0001	0.0001	5
	1993	0.003	0	4	0.003	0.00001	4	0.0030	0.00001	5	0.0003	0	4	0.0003	0	4	0.0003	0	5

3. Macrophytes in Island Lake show shifts in community structure consistent with increases in salinity. TAEM (1994) using linear lake transects showed that 11 species were eliminated, and that 5 were new. In shoreline transects they found 4 species previously not described, and they found that 4 species were eliminated. Population increases were found in a number of salt tolerant species. Only a few permanent riparian/aquatic stations have been established by the company in the fen downstream of Island Lake to assess long term changes in species abundance and density.

4. Major changes in the benthos of Island Lake are typical of locations with severely degraded water quality. Both Swanson (1991) and Hynes (1990) suggested the changes in the benthic community of Island Lake were principally caused by increased salinity rather than by high concentrations of metals and radionuclides.

   For Snake Lake benthos, the similarity between the community composition and diversity found in 1993 and baseline, suggests that impacts in Snake Lake are not severe.

   Minor effects may be occurring in the north end of Cluff Lake. These consist of a slightly lower species richness and reduced diversity.

   Stream data are much less comparable and impact analyses are inconclusive because of problems such as changes in timing of benthos collections and lack of similarity of methods.

5. Major changes in fish density and biomass have occurred in Island Lake and may indicate reduced predation pressure on the white sucker population from northern pike which are intolerant of conditions in the lake.

   Conversely, a six fold increase in northern pike biomass in Snake Lake was detected in 1993 compared to 1978 baseline data. The reason for this increase is not apparent.

   Large fish in Cluff Lake and Sandy Lake do not appear to have undergone any major shifts in community structure or biomass.

6. Elevated concentrations of uranium, molybdenum, arsenic and Ra-226 have been found in macrophytes from Island Lake, and uranium and Ra-226 have been found at higher concentrations in macrophytes in Boulder Creek. Concentrations peaked in Boulder Creek macrophytes during 1983-84. Increases were not detected in macrophytes downstream in Cluff Lake.

   Analyses of fish from Cluff Lake, Sandy Lake and Island lake show that only fish from Island Lake have bioaccumulated uranium and molybdenum.

1.5 Toxicity Testing

Historically, a battery of tests have been used to evaluate the toxicity of effluent, particularly between 1983 and 1987. During that period tests included:

- rainbow trout (96-h acute static)
- *Daphnia magna* (48-h acute static), *D. magna* and *D. pulex* (96h EC_{50})
- *Selenastrum* (growth)
- Microtox (luminescence)
- *Stizostedion vitreum* (6.4 cm long, 96h static, done in 1984)

Since 1988, 96h static tests with rainbow trout have been conducted once or twice per year.

There has been no evidence of ongoing acute toxicity. However, some of the other tests on *D. magna*, and *D. pulex* and *Selenastrum* have shown acute toxicity, usually related to process upsets (Swanson 1991).

Hynes (1990) reviewed toxicity testing and found correlations between observed toxicity and iron and that uranium was associated with toxicity for *Selenastrum* and *Daphnia*.

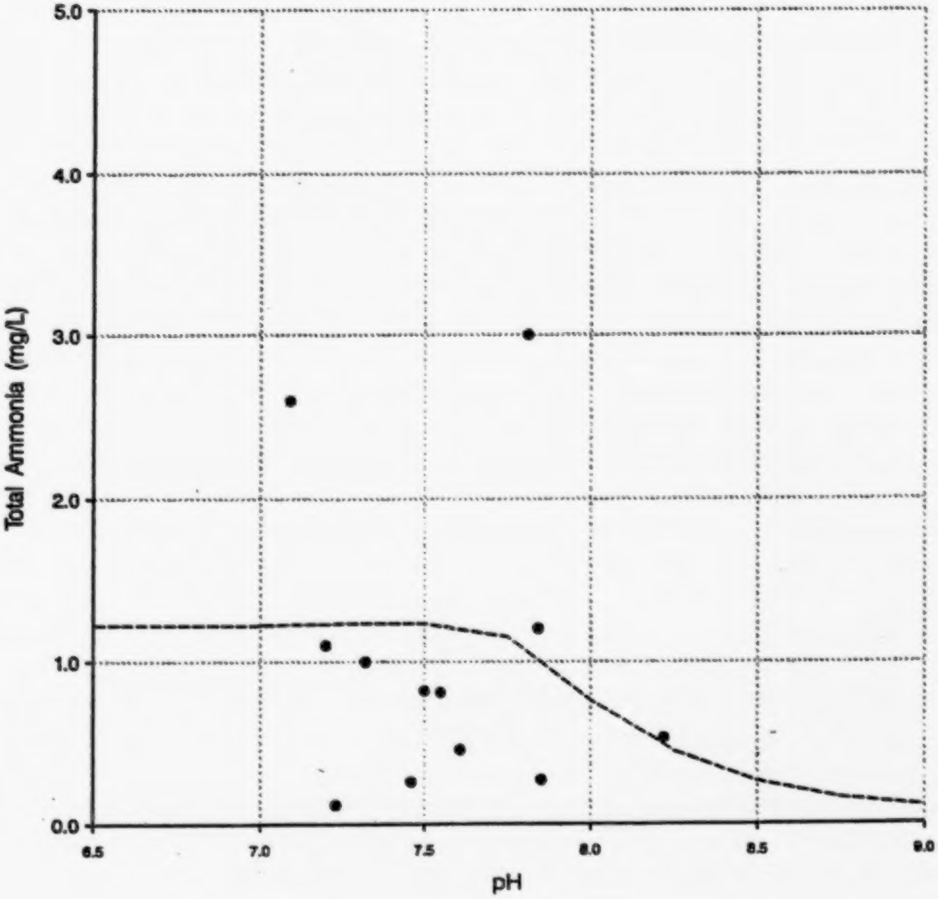
Hydroqual (1992) also found no evidence of acute toxicity in the rainbow trout test, *Daphnia*, Microtox, or the fathead minnow mortality test. However, there was some evidence of chronic effects on *Ceriodaphnia dubia* based on mortality and fecundity end points, and on nematode *Panagrelus sp.* growth. Hydroqual cited problems with sensitivity of *Ceriodaphnia* for testing effluents having wide ranging alkalinities.

The toxicity of total ammonia nitrogen (NH_3-N and NH_4-N) is related to the amount of un-ionized ammonia (NH_3-N) in solution. This is modified largely by temperature and pH. Both acute (B.C. Environment 1986) and chronic toxicity (SSWQO 1988) objectives are available which allow the determination of levels of total ammonia above which un-ionized ammonia may be harmful to aquatic life such as fish.

Water temperatures have not been recorded in the Cluff Lake operation environmental database. Temperature data from TAEM (1994) collected in May and August 1993 in Island Lake ranged from 12.4 to 12.8°C in May to 16.5 to 19.8°C in August. In an attempt to predict if ammonia levels in Island Lake water have, historically, been potentially toxic, Figure 1.17 presents a scatter plot of total ammonia versus pH levels. The dotted line in the figure represents the Saskatchewan surface water quality objective for total ammonia, at a temperature of 20°C. Values of total ammonia which plot above this line may be potentially toxic to aquatic life.

Assuming a worst case water temperature of 20°C, most of the 24 recorded ammonia-pH combinations in Figure 1.17 result in un-ionized ammonia levels below the potentially toxic level. Two points clearly lie above SSWQO objectives. Two other points may exceed SSWQO objectives, and were recorded in June when warmer water temperatures are expected.

Figure 1.17: Total ammonia versus pH levels in Island Lake. Black dotted line represents surface water quality objective for total ammonia, at various pH levels, with water temperature at 20°C.



In consideration of the toxicity record for the Cluff Lake Mine, both Hynes (1990) and Swanson (1991) suggested that toxicity testing using a battery of organisms at different levels of organization should be used to test for downstream chronic/sublethal long term effects. Hydroqual (1992) recommended that species found naturally in lakes near uranium mines should be chosen for further chronic tests and that work should be done on establishing the relative contributions of salinity, metals and ammonia to toxicity. This would require protocols such as adjusting the salinity of diluent and controls, and appropriate acclimation of test organisms.

1.6 Summary of Effects

The Cluff Lake operation has a consistent record of compliance with the MMLER as well as other provincial license release limits. Concentrations of all radionuclides and metals, as well as totals suspended solids, and pH have generally been well below limits. However, effects have occurred (per original predictions) in the mixing zone. Concerns have been raised regarding downstream cumulative effects due to elevated concentrations of salinity, ammonia, uranium and possibly molybdenum.

In general, a review of surface water quality shows conclusively that parts of the system that receive effluent from the mine (i.e. the Island Lake drainage) have elevated concentrations of uranium, molybdenum, and major ions (calcium, magnesium, potassium, sodium) and sulphate. These have resulted from discharge from the STS into Snake Creek and seepage from the TMA into Snake Lake. Uranium, molybdenum and nickel concentrations have been elevated in sediments within this system downstream as far as Sandy Lake. These increases in contaminants are also reflected in higher concentrations of uranium, molybdenum, arsenic and radium-226 in macrophytes from Island Lake and the fen downstream, and higher uranium and molybdenum concentrations in fish. The fen downstream of Island Lake apparently protects downstream areas of Island Creek because most (96%) of the elements discharged from the STS are retained at that point (Swanson, 1990).

Major changes in composition of the macrophyte, fish, benthic, and zooplankton communities have been demonstrated within Island Lake; however, there is little evidence to confirm changes in other parts of the watershed. In some cases this is due to insufficient spatial and temporal replication, changes in sampling methods causing incomparable data sets, or simply absence of sufficient population size for continued sampling. Some effects of physical disturbances from construction activities on sediments and water have been detected in Peter River and Earl Creek and some indication of increases in concentrations of metals and radionuclides in north Cluff Lake have been reported. Drainage from waste rock and a cement batch plant are probable sources of contamination in this watershed.

1.7 Summary of Recommendations

- salinity has been identified as the probable cause of effects on aquatic biota in Island Lake
    - salinity should be considered for inclusion in the revised MMLER, recognising that it is largely specific to the uranium mining industry
    - the use of alternate technology or chemicals used in milling or effluent treatment systems should be assessed as a possible means of reducing problems associated with high salinity. The need for such low radium-226 levels in the effluent should be assessed, since achieving these levels requires significant amounts of reagent. However, reduction of reagents as a possible method of lowering salinity needs to be balanced with radiological risk to the receiving watercourse. The later strategy is supported by Hynes (1990) who concluded that "we have been able to achieve lower and lower concentrations of radium in the effluent discharge, but at the cost of considerably increased reagent loading to the effluent. The extreme focus on radium has also detracted attention from contaminants more likely to be of concern..."

- uranium should be considered for inclusion in the revised MMLER, recognising that it is largely specific to the uranium mining industry

- molybdenum is increasing or accumulating downstream in the Island Lake drainage
    - molybdenum should be considered for inclusion in the revised MMLER

- A consistent record of both field pH and temperature are required with total ammonia data to monitor compliance
    - ammonia should be considered for inclusion in the revised MMLER

- work needs to be done on establishing the relative contribution of salinity, metals and ammonia from the effluent to downstream impacts. To this end, the use of chronic toxicity testing using a battery of test organisms has been recommended (Swanson 1991), and could be used a method of determining the potential extent of biological effects in both the Island Creek and Cluff Lake drainages. These tests should consider the use of species occurring near uranium mines (or appropriate sensitive surrogates) and should include tests on both water and sediments.

- there has been an overwhelming amount of work conducted to date on Island Lake, which is clearly impacted.
    - Less detailed work could now monitor changes in these locations.
    - This would permit a focusing of monitoring into more accurate determination of the spatial extent of biological effects, including determining how far downstream of

Island Lake effects occur, and the extent of effects in the Cluff Lake system.

- Focusing studies on a few key biological endpoints would probably result in improved study designs.
  - Swanson (1991) reviewed the monitoring being conducted and suggested that the macrophyte, benthos and fish community surveys be continued.
  - Benthic surveys could be improved by standardizing sampling methods and timing, and by improving on spatial and temporal reference information. Survey results should be evaluated using both conventional and multivariate data analysis methods which are more appropriate and sensitive when working with data from community surveys (DFO and EC 1995).

- Improvements to the design and assessment of fish populations may be possible by examining small and large fish species community structure, population characteristics, chemical body burden data, and use of biochemical indicators to assess the extent and magnitude of effects on ecosystem health (Munkittrick and Dixon 1989).

- TAEM and SENES (1995) recommend continued monitoring for bioaccumulation in aquatic macrophytes in Island Lake, and for fish in Cluff, Sandy, and Island Lake. The continuation of these monitoring programs is important since it allows trend assessment of bioaccumulation over time.

- TAEM and SENES (1995) concluded that for First to Fourth Lakes south of Cluff Lake, there are no indications of mining related effects. The lack of connectivity of these lakes to mining activities indicates that their continued monitoring should be reassessed.

- Appropriate quality assurance programs are needed to ensure long term usefulness of data collected
  - There should be a method to ensure greater return to site accuracy since this has been indicated as a possible source of error.

1.8 References

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Appendix 1: Water Quality Monitoring License Requirements

RECEIVING WATERS

SAMPLE LOCATION	PARAMETERS 2,3	FREQUENCY
1.5 Cluff Lake (at outlet)	Class A Water analyses	A
3.1 Snake Lake (near outlet)	Water level	W,7
	Class E water analyses	M
	Class A water analyses	SA
3.2 Island Lake Near Outlet 3.2.5 Island Creek at Dolomites	Class E water analyses, Mo, Al	M
3.3 Island Creek at Sandy Lake	Class E water analyses, Mo, Al	Q
3.4 Douglas River at Inlet to Sandy Lake 3.5 Douglas River at Outlet of Sandy Lake	Class A water analyses	SA

SECONDARY TREATMENT SYSTEM (S.T.S.) EFFLUENT

SAMPLE LOCATION[1]	PARAMETERS[2,3]	FREQUENCY[4]
4.0 Secondary Treatment System Discharge to Snake Creek - Composite Sample (5)	Class E water analyses Ba, Al	W, 5
	As, Cu, Pb, Mo, Ni, U, Zn, Ra-226, Th-230, Pb-210, Po-210, TSS, NH_3-N, Al, Ba	M,5
	Class A water analyses	Q,5
	Volume of water treated and released; Kg chemicals used; (BaCl2, $AlSO_4$)	Monthly

CLAUDE & DJ OREBODY

SAMPLE LOCATION[1]	PARAMETERS[2,3]	FREQUENCY[4]
5.2 Claude Creek at Outlet of Claude Lake	Class C water analyses	SA

OTHER

SAMPLE LOCATION[1]		PARAMETERS[2,3]	FREQUENCY[4]
6.0	Peter River Off North End of Airstrip	Class C water analyses	Quarterly
6.1	Peter River at Mill Road Crossing	Class C water analyses	April, July, November
6.1.5	Earl Creek at Airport Road Crossing	Class C water analyses	S.A.
6.2	Earl Creek at Mill Road Crossing		
6.3	Pumped Water to Snake Lake From Tailings Management Area Fresh Water Diversion Dam, East Side	Class E water analyses Dates pumping, duration, volume pumped	May, August, October
6.4	Pumped Water to Snake Lake From Tailings Management Area Saddle Dam, West Side		

1. Locations shown on attached Figure 1.

2. pH and conductivity measured both in the field and in the laboratory.

3. Parameters:

   - Class A - Total sample for - HCO_3/CO_3 [6], Cl, Ca, Al, Mg, K, Na, SO_4, NH_3-N, Total Nitrogen, Total P, TSS, Pb-210, Ra-226, Po-210, Th-230, U, As, B, Hg, Se, Bi, Cd, Co, Cr, Cu, Fe, Mn, Ni, Pb, Zn, Ba, Mo, V, pH, Conductivity, Total Hardness, Sum of Ions, Turbidity.

   - Class C - Total sample for - HCO_3/CO_3 [6], Ca, Cl, Mg, K, Na, SO_4, TSS, Ra-226, U, As, Cu, Fe, Ni, Zn, Pb, pH, Conductivity, Total Hardness, Sum of Ions, Turbidity.

   - Class D - Total sample for - TSS, Ra-226, U, As, Cu, Fe, Ni, Zn, Pb, pH, Conductivity, Total Hardness, Turbidity.

   - Class E - Total sample for HCO_3/CO_3 [6], Ca, Cl, Mg, K, Na, SO_4, TSS, Ra-226, U, pH, Conductivity, Sum of Ions, Turbidity.

4. Frequency
   - D - once per day
   - W - once per week
   - M - once per month
   - TM - twice monthly
   - Q - every three months
   - SA - once every six months
   - A - once per year (September)

5. Composite Sample:

   A 24-hour composite sample is to be collected using a continuous automatic sampler. However, in the event of a failure of this device, samples are to be collected by manual methods and are to consist of four equal volume collected at approximately equal time intervals over a twelve hour period. The reported results shall stipulate which method was used.

6. HCO_3/CO_3 - When the pH is greater than 8.3, report HCO_3 and CO_3. When the pH is less than 8.3 report HCO_3.

7. During ice free months.

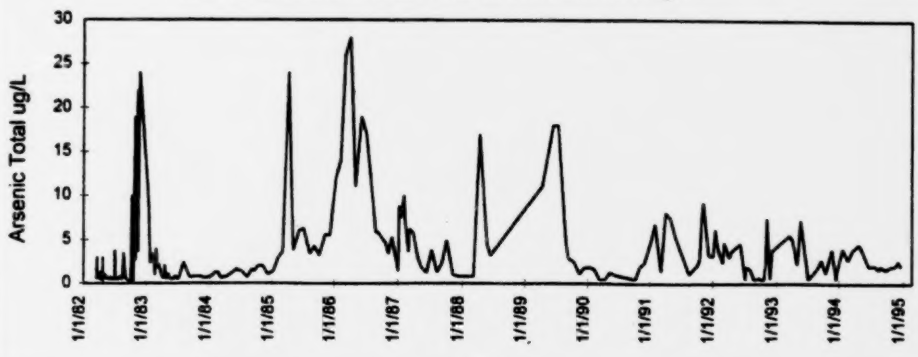
8. Monthly composite.

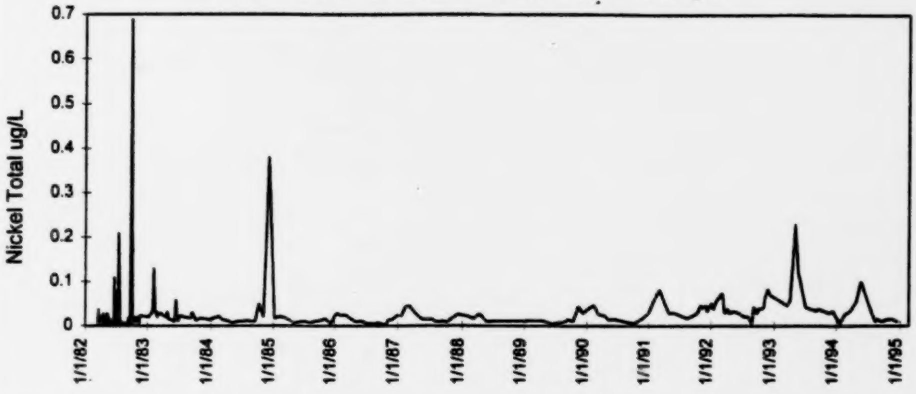
9. Sample to be collected during spring runoff or rainstorm events.

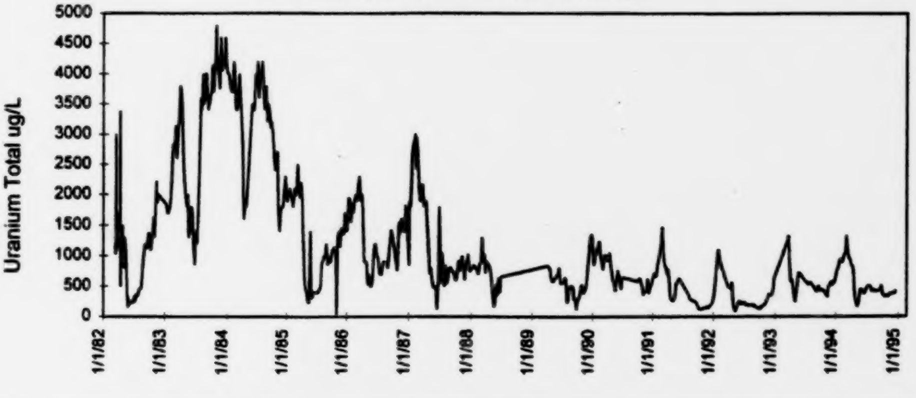
10. Analyses to include analysis for Total and Dissolved Ra-226, U, As and Ni.

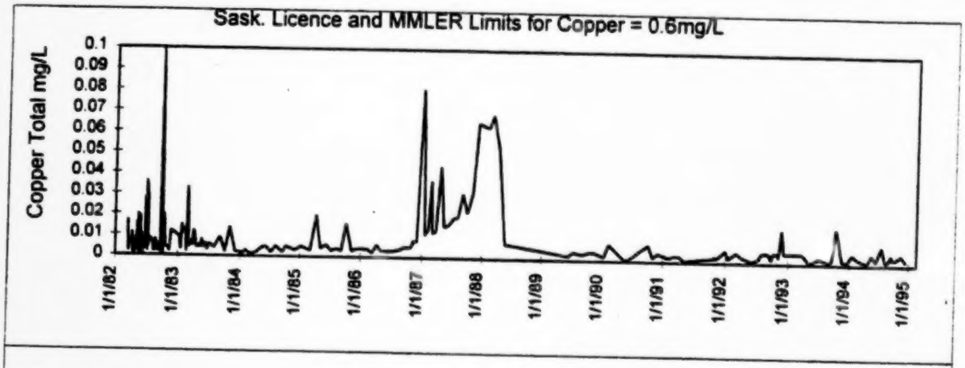
11. Pit to be sampled once access is safe to do so.

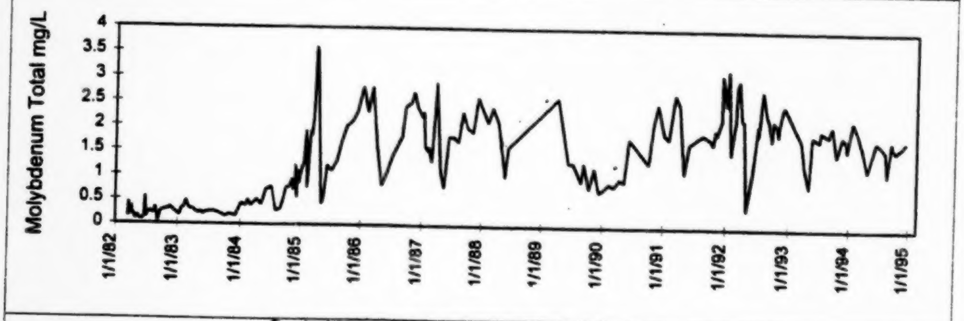
Appendix 2: Quality of Final Effluent Discharge from Secondary Treatment System

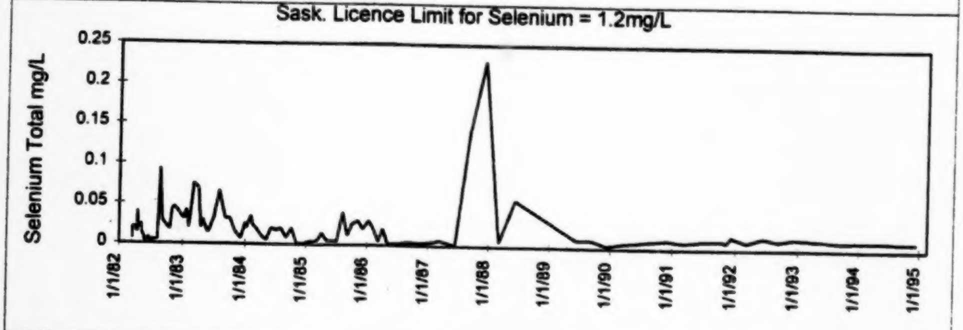

Sask. Licence and MMLER Limit for Arsenic = 1000 ug/L

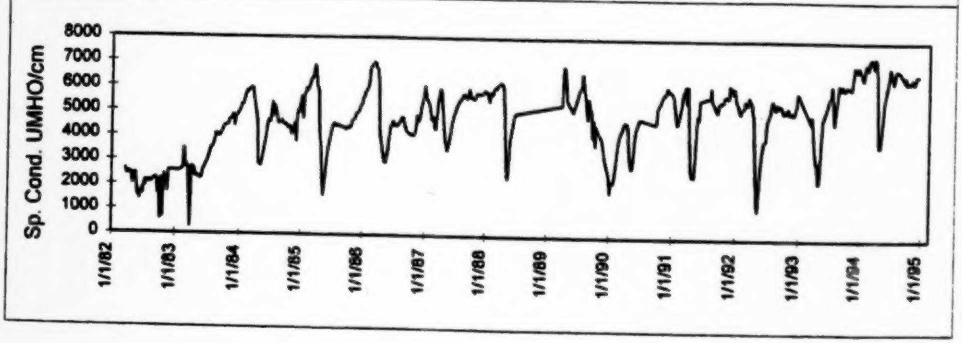

Sask. Licence and MMLER Limit for Nickel = 1000 ug/L

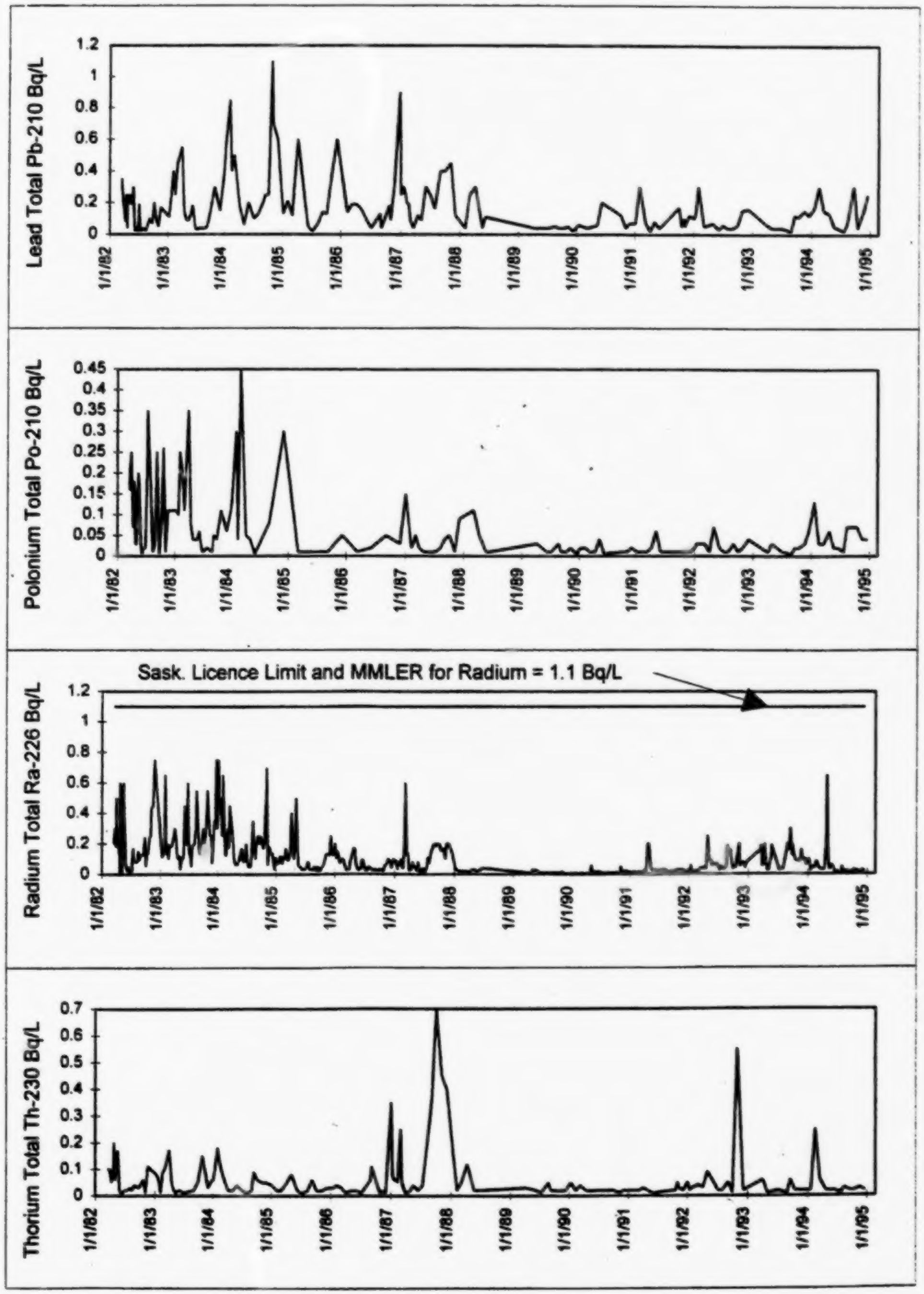

Sask. Licence Limit for Uranium = 5000 ug/L

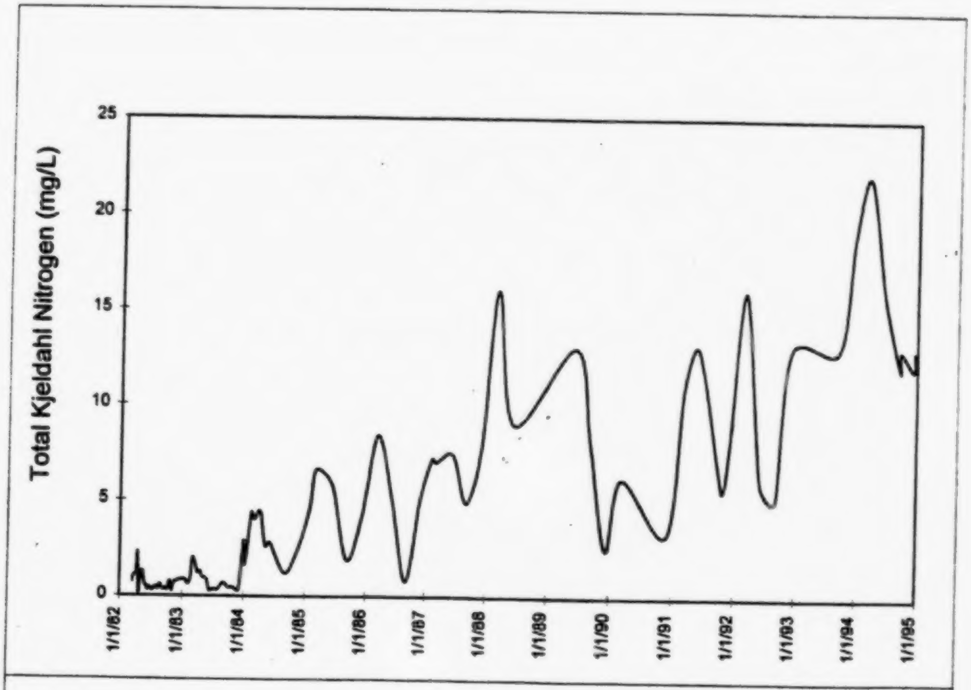

Sask. Licence and MMLER Limits for Copper = 0.6mg/L
Copper Total mg/L
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95

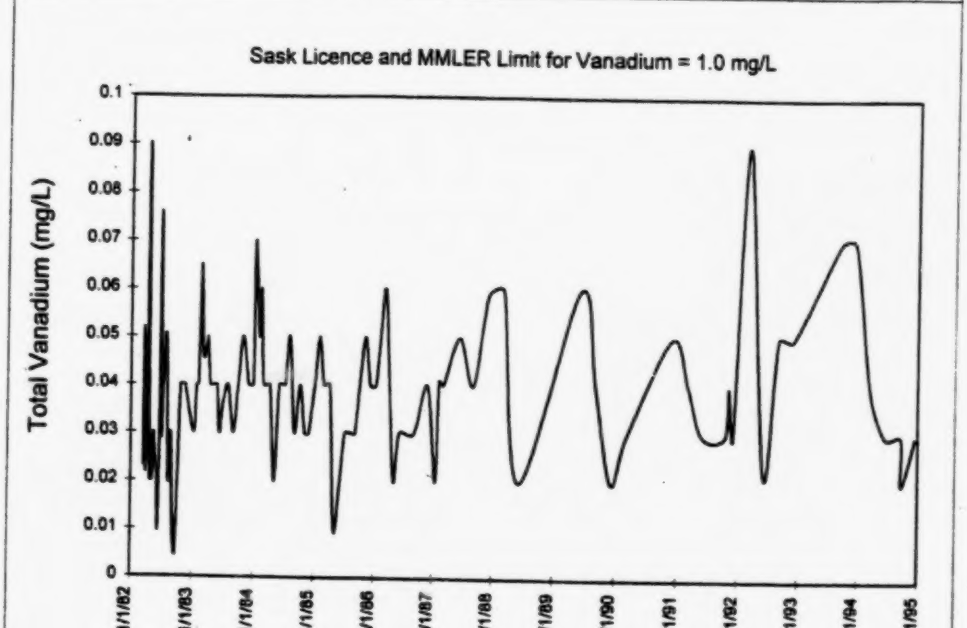

Molybdenum Total mg/L
4
3.5
3
2.5
2
1.5
1
0.5
0
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95

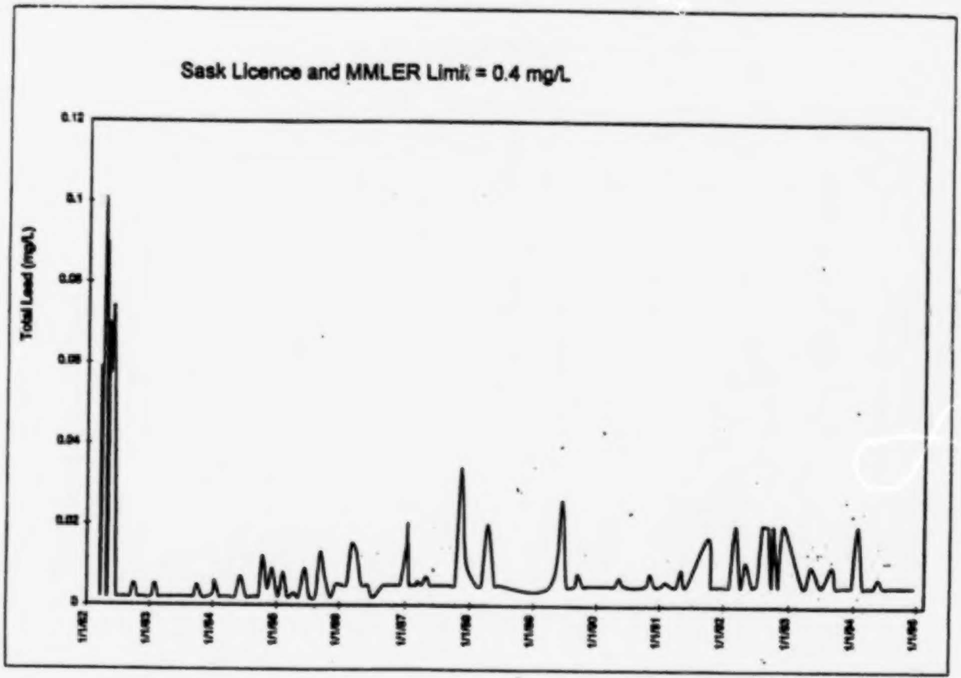

Sask. Licence Limit for Selenium = 1.2mg/L
Selenium Total mg/L
0.25
0.2
0.15
0.1
0.05
0
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95


Sp. Cond. UMHO/cm
8000
7000
6000
5000
4000
3000
2000
1000
0
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95

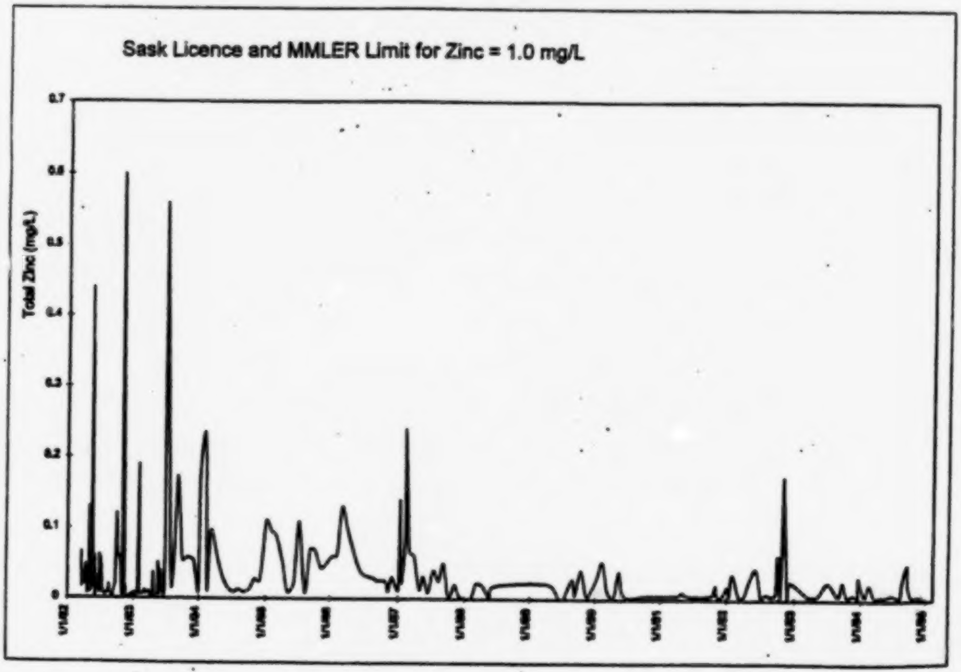

Lead Total Pb-210 Bq/L
1.2
1
0.8
0.6
0.4
0.2
0
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
Polonium Total Po-210 Bq/L
0.45
0.4
0.35
0.3
0.25
0.2
0.15
0.1
0.05
0
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
Sask. Licence Limit and MMLER for Radium = 1.1 Bq/L
Radium Total Ra-226 Bq/L
1.2
1
0.8
0.6
0.4
0.2
0
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
Thorium Total Th-230 Bq/L
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95

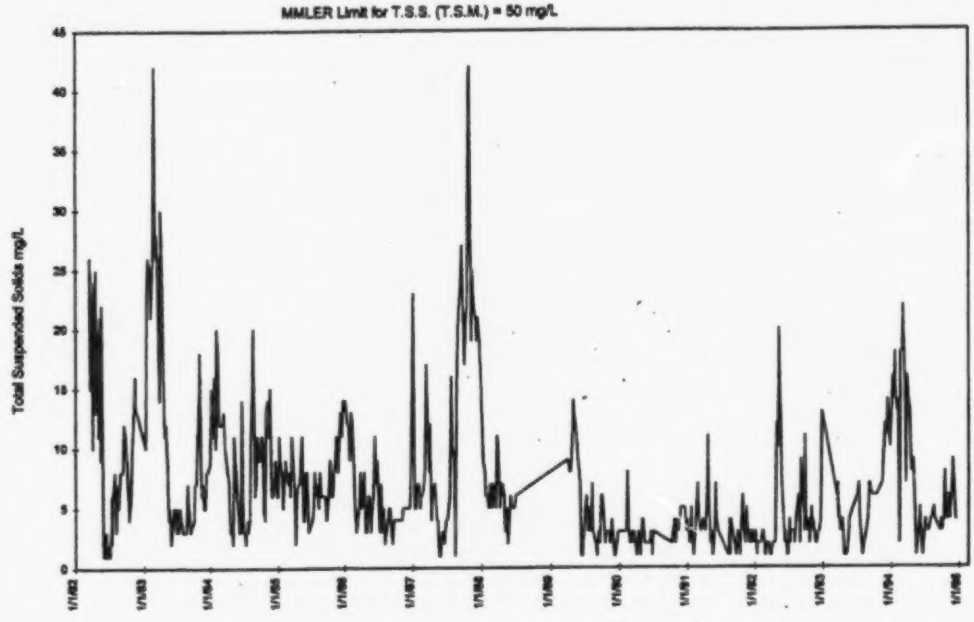



Sask Licence and MMLER Limit for Vanadium = 1.0 mg/L

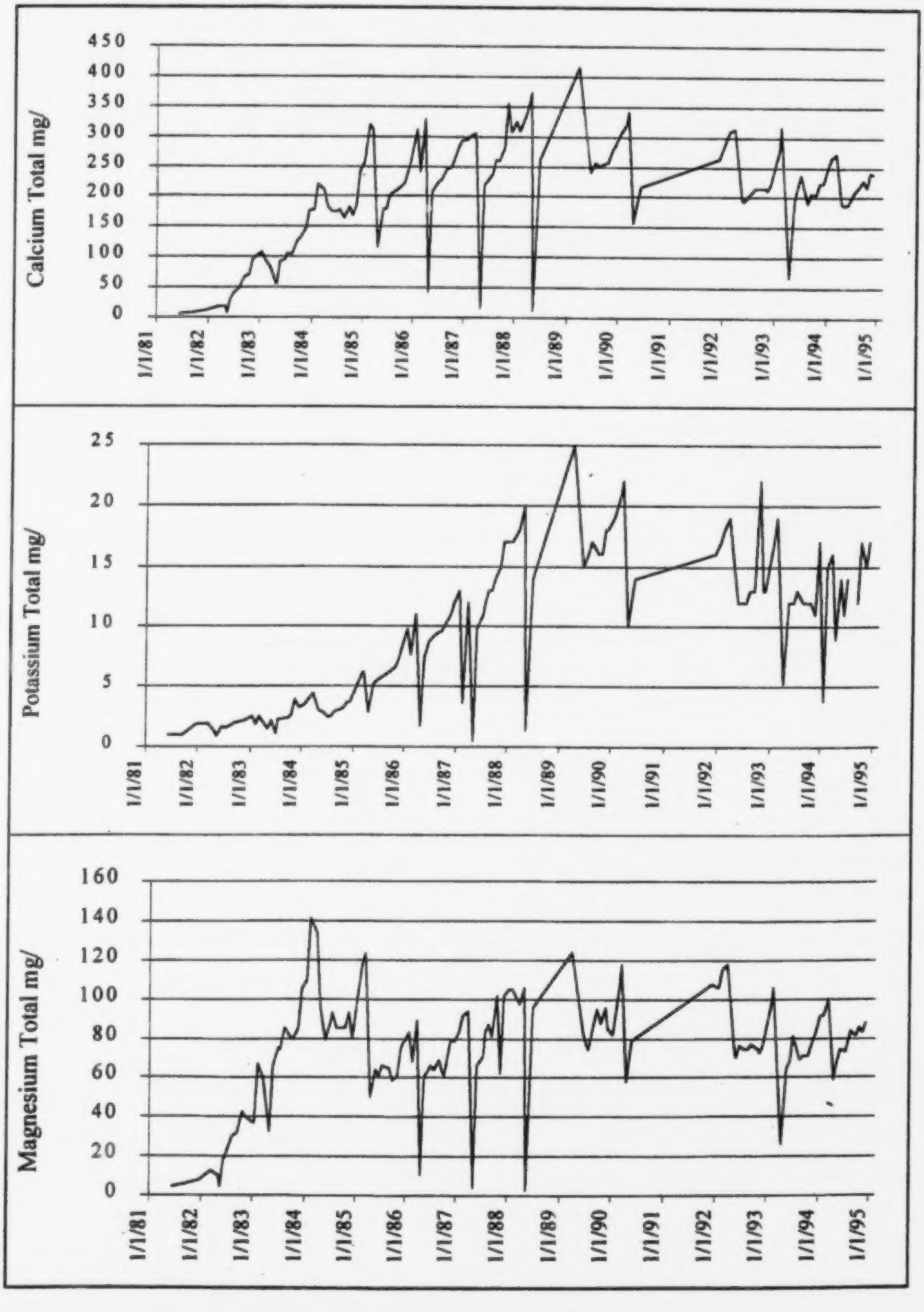


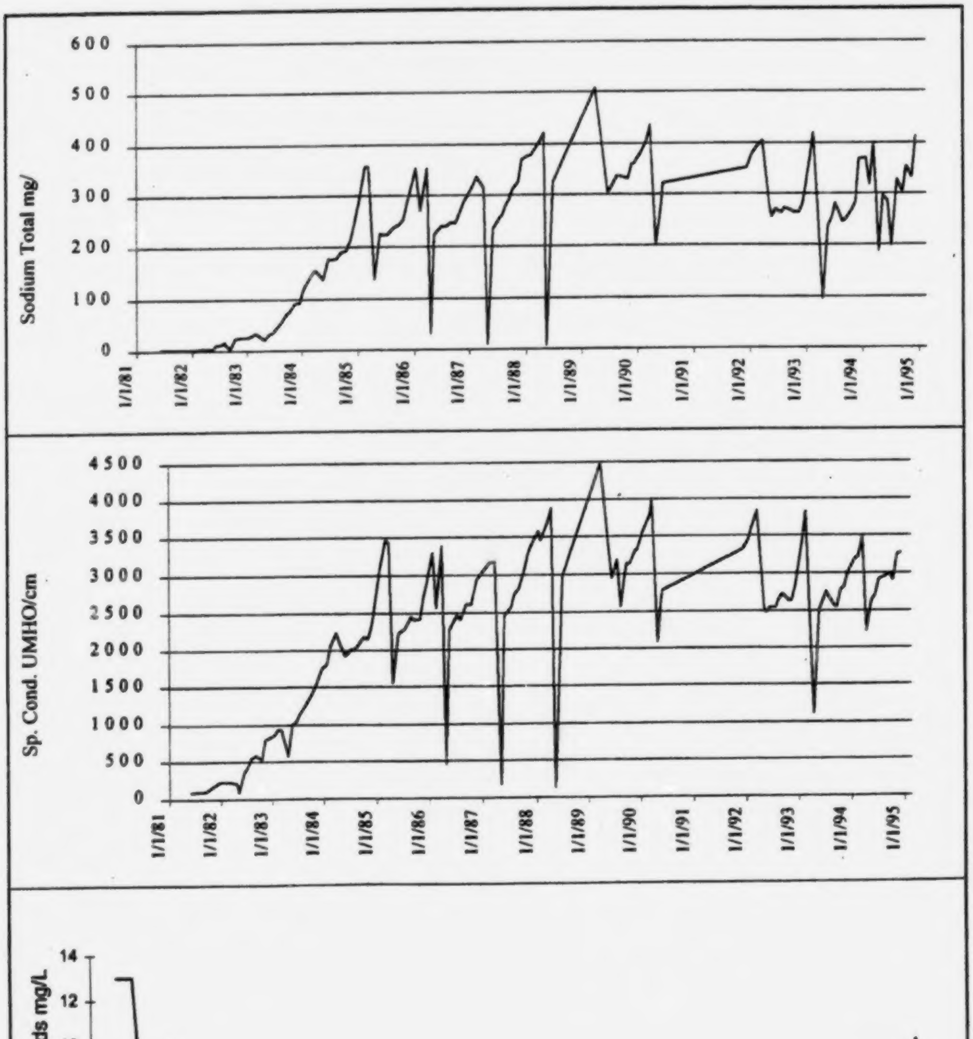
Cogema Cluff Lake Final Effluent Discharge from STS

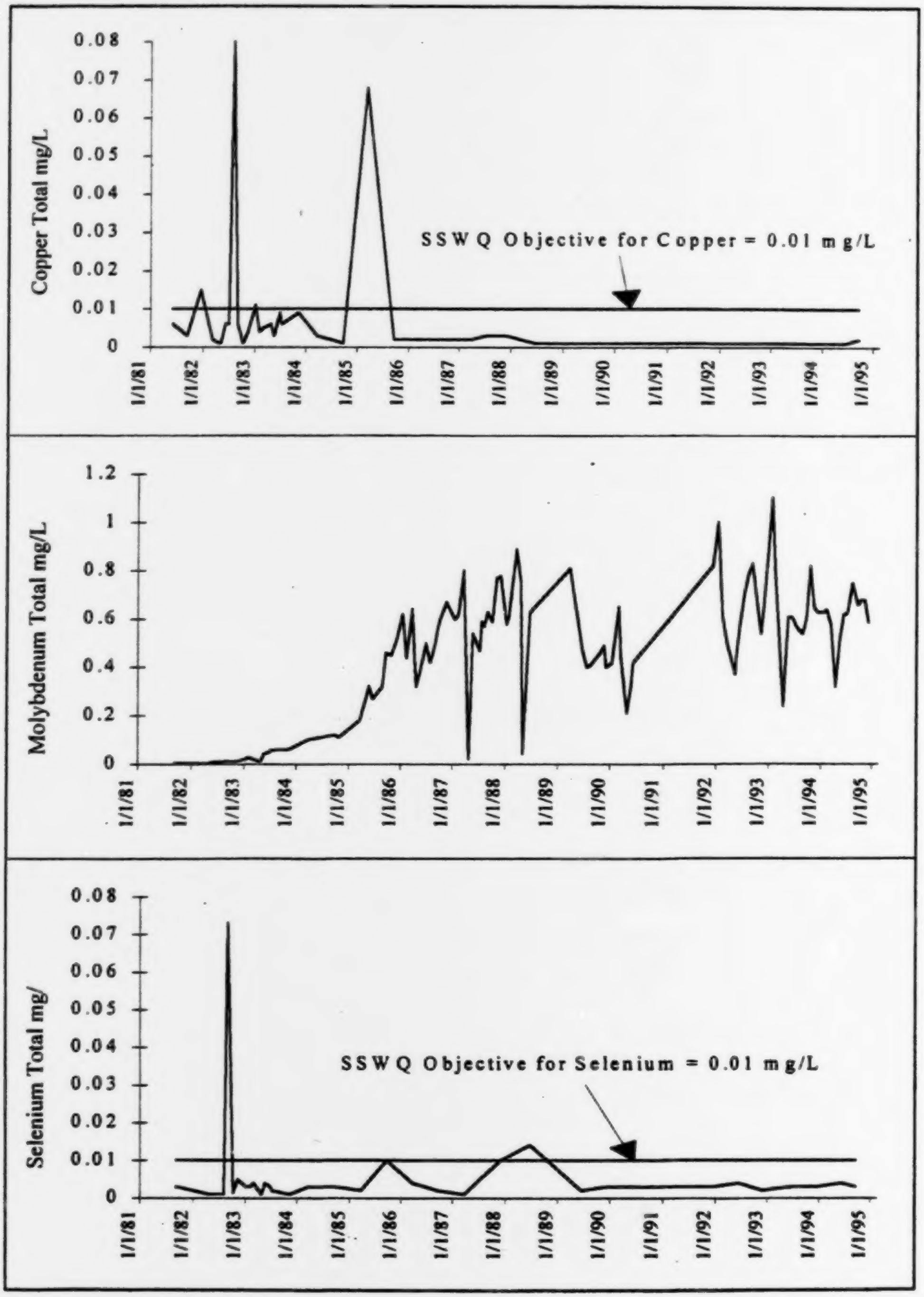


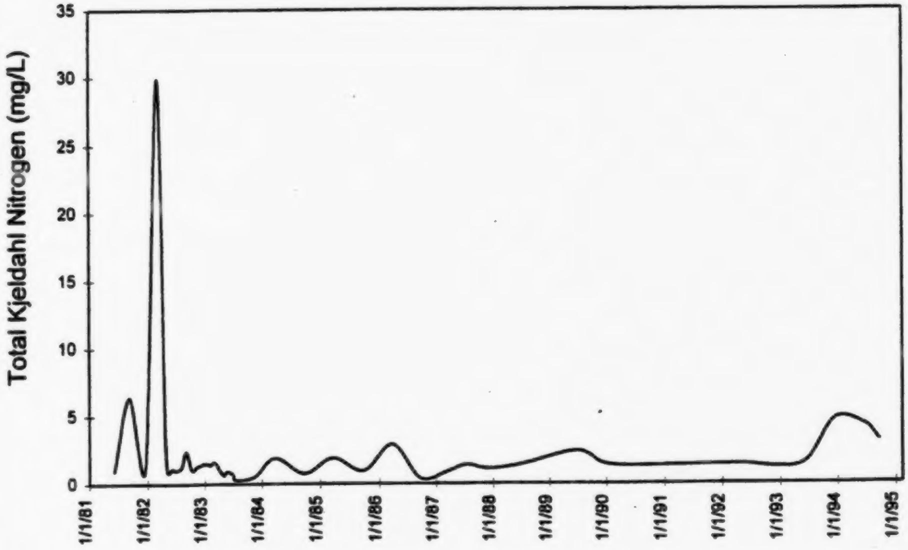

MMLER Limit for T.S.S. (T.S.M.) = 50 mg/L
Total Suspended Solids mg/L
0
5
10
15
20
25
30
35
40
45
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95

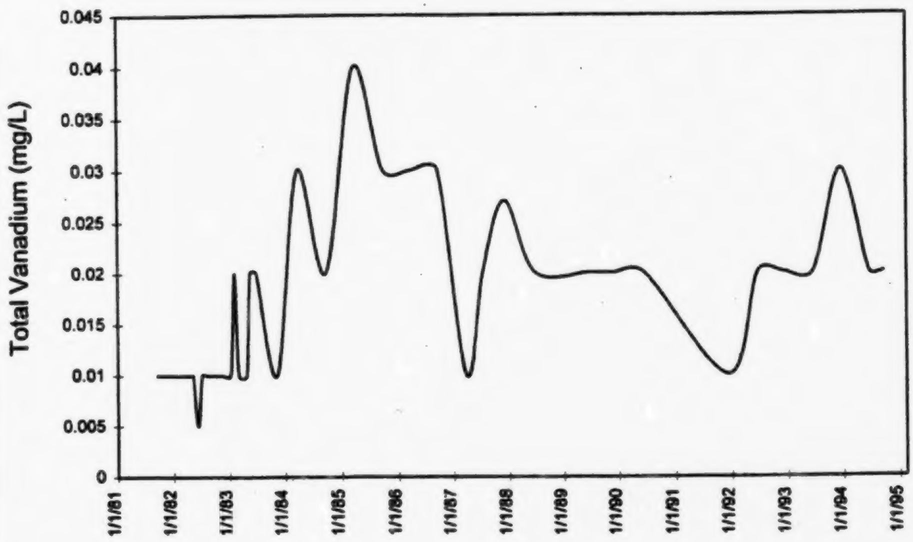
Appendix 3: Water Quality Data for Discrete Samples from Island Lake

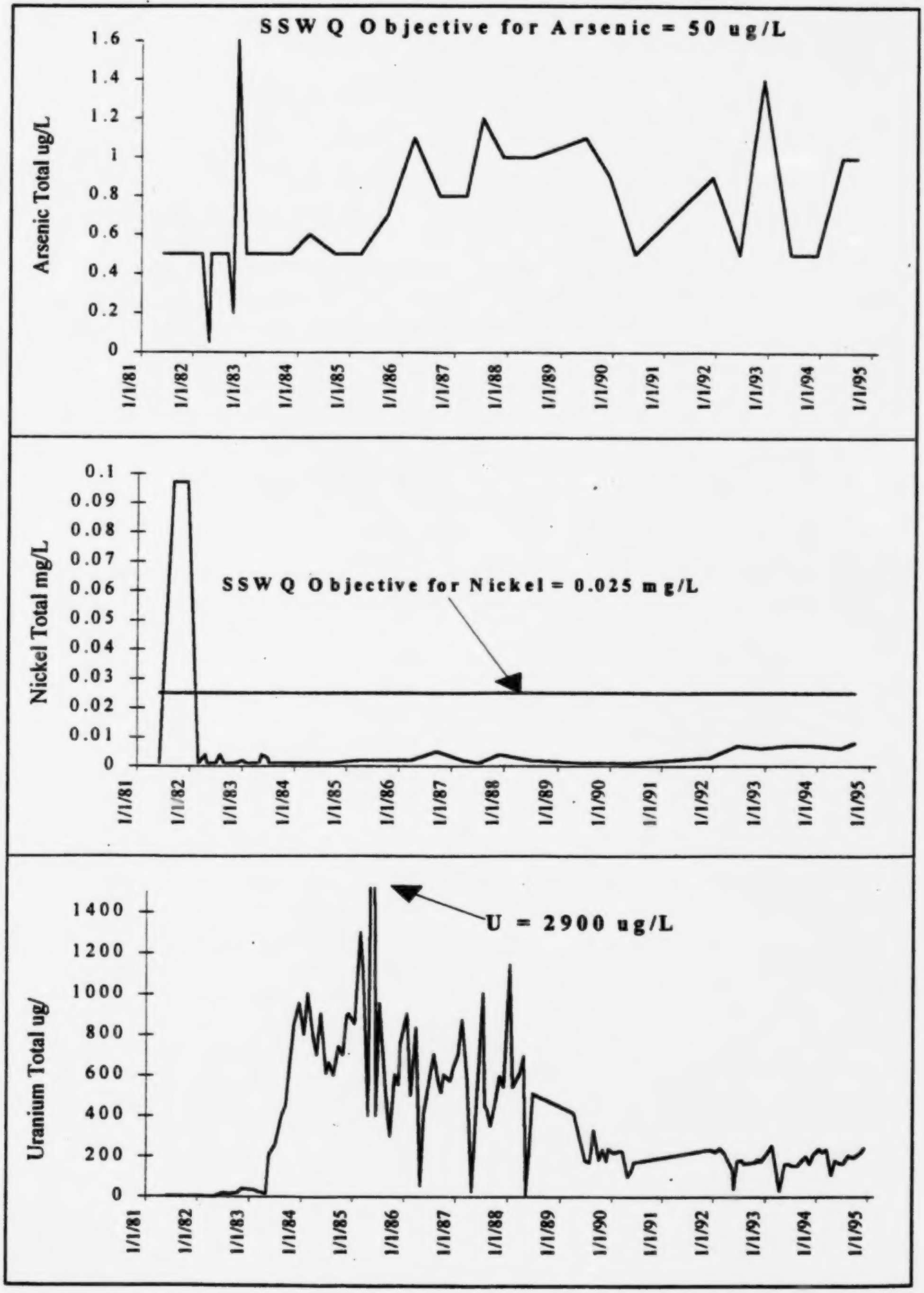



Sodium Total mg/
600
500
400
300
200
100
0
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
Sp. Cond. UMHO/cm
4500
4000
3500
3000
2500
2000
1500
1000
500
0
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
Total Suspended Solids mg/L
14
12
10
8
6
4
2
0
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95


0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
Copper Total mg/L
SSWQ Objective for Copper = 0.01 mg/L
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
1.2
1
0.8
0.6
0.4
0.2
0
Molybdenum Total mg/L
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
Selenium Total mg/
SSWQ Objective for Selenium = 0.01 mg/L
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95




SSWQ Objective for Vanadium (Livestock Protection) = 0.1mg/L


SSWQ Objective for Arsenic = 50 ug/L
Arsenic Total ug/L
1.6
1.4
1.2
1
0.8
0.6
0.4
0.2
0
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
Nickel Total mg/L
0.1
0.09
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
SSWQ Objective for Nickel = 0.025 mg/L
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
Uranium Total ug/
1400
1200
1000
800
600
400
200
0
U = 2900 ug/L
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
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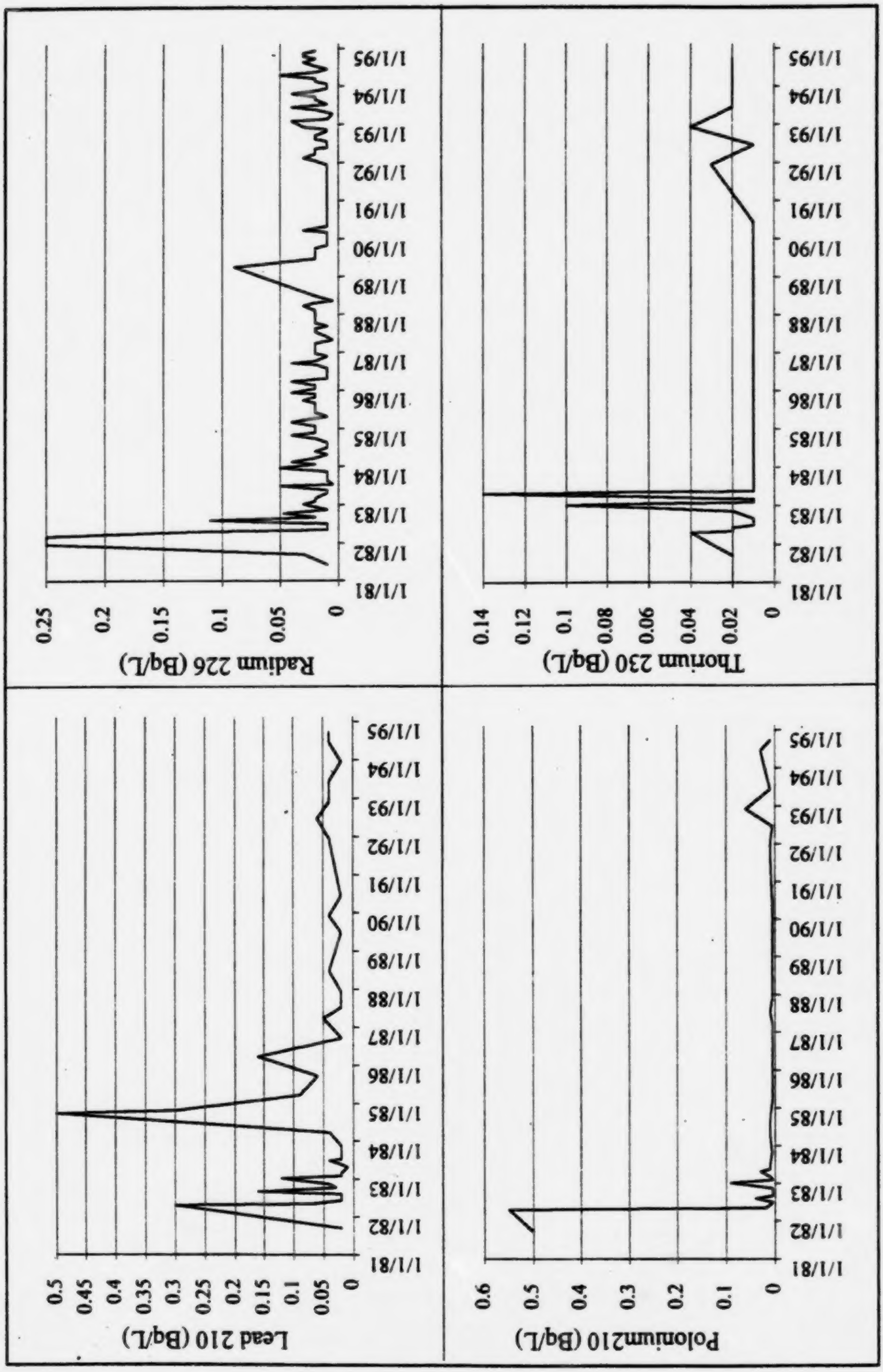
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95


Radium 226 (Bq/L)
0.25
0.2
0.15
0.1
0.05
0
Thorium 230 (Bq/L)
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
Lead 210 (Bq/L)
0.5
0.45
0.4
0.35
0.3
0.25
0.2
0.15
0.1
0.05
0
Polonium210 (Bq/L)
0.6
0.5
0.4
0.3
0.2
0.1
0
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95

Appendix 4: Surface Water Quality Objectives, Saskatchewan Environment and Public Safety, November 1988

3.1 General Objectives for Effluent Discharges

The following basic objectives are applicable to all waters receiving effluents, **including the mixing zones adjacent to effluent outfalls** (see Section 3.1) in the context that municipal. industrial, agricultural and other discharges should be:

(1) free from substances in concentrations or combinations which are acutely toxic or may be harmful to human, animal or aquatic life;

(2) free from substances that will settle to form putrescent or otherwise objectionable sludge deposits, or that will adversely affect aquatic life or waterfowl;

(3) free from debris, oil, grease, scum or other materials in amounts sufficient to be noticeable in the receiving water;

(4) free from colour, turbidity or odour-producing materials that would:

(a) adversely affect aquatic life or waterfowl;
(b) significantly alter the natural colour of the receiving water;
(c) directly or through interaction among themselves or with chemicals used in water treatment; result in undesirable taste or odour in treated water, and;

(5) free from nutrients in concentrations that create nuisance growths of aquatic weeds or algae or that results in an unacceptable degree of eutrophication of the receiving water;

(6) in addition to the above objectives, effluent discharged to surface waters should not utilize more than 30 percent of the assimilation capacity of the receiving waterbody when discharged via means of a diffused outfall, or more than 10 percent when discharged via a point source outfall. These design objectives should be utilized during the planning stages of projects involving effluent discharges. For purposes of determining the available assimilation capacity of a receiving waterbody a flow rate equal to or less than the average seven day low flow which occurs once in ten years, at the outfall area generally should be used.

TABLE 4.2

SPECIFIC SURFACE WATER QUALITY OBJECTIVES FOR THE PROTECTION OF AQUATIC LIFE AND WILDLIFE

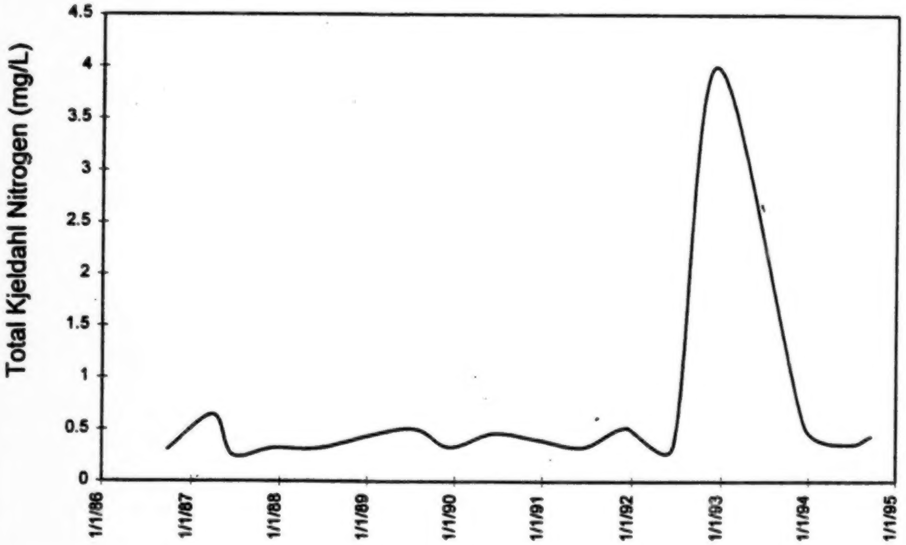
(values are in mg/L unless otherwise stated)

Arsenic (total)	0.05
Barium (total)	1.0
Cadmium (total)	0.001
Chlorine (total residual)	0.002
Copper (total)	0.01
Cyanide (total)	0.01
Iron (total)	1.0
Lead (total)	0.02
Mercury (total)	0.0001
Nickel (total)	0.025 mg/L where hardness $\leq$ 100 mg/L ($CaCO_3$) 0.100 mg/L where hardnes $>$ 100 mg/L ($CaCO_3$)
Oxygen (dissolved)	6.0 - Protection of early life stage of warm water biota 5.0 - other life stages of warm water biota 9.5 - early life stages of cold water biota 6.5 - other life stages of cold water biota
Selenium (total)	0.01
Silver (total)	0.01
Temperature	not to be altered by more than 3°C from ambient water temperature
Zinc (total)	0.05
Lindane (γBHC)	0.0001
Pentachlorophenol	0.0005
Phenols (total)	0.001
Phenoxy herbicides (2,4-D)	0.004
Polychlorinated biphenyls (total)	0.000001
Unspecified substances	To provide reasonably safe concentrations of substances not specifically addressed in the above objectives, a general objective of 1/100 of the 96 hour LC_{50} of such substances shall be applied. Consideration will also be given to such effects as objectionable taste and odour generation and tainting flesh of aquatic life.

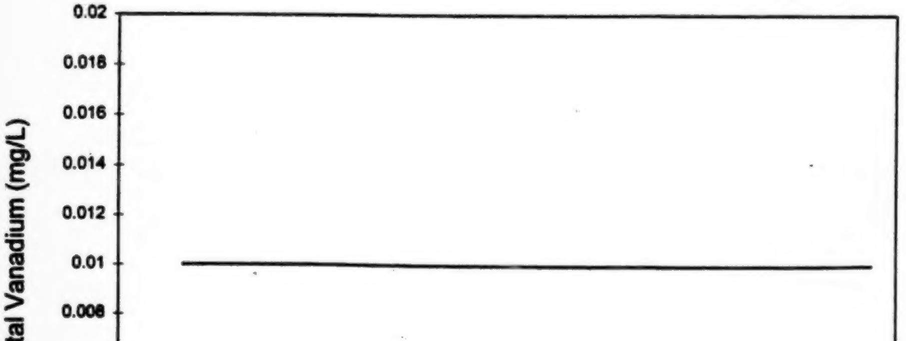
General Surface Water Quality Objectives

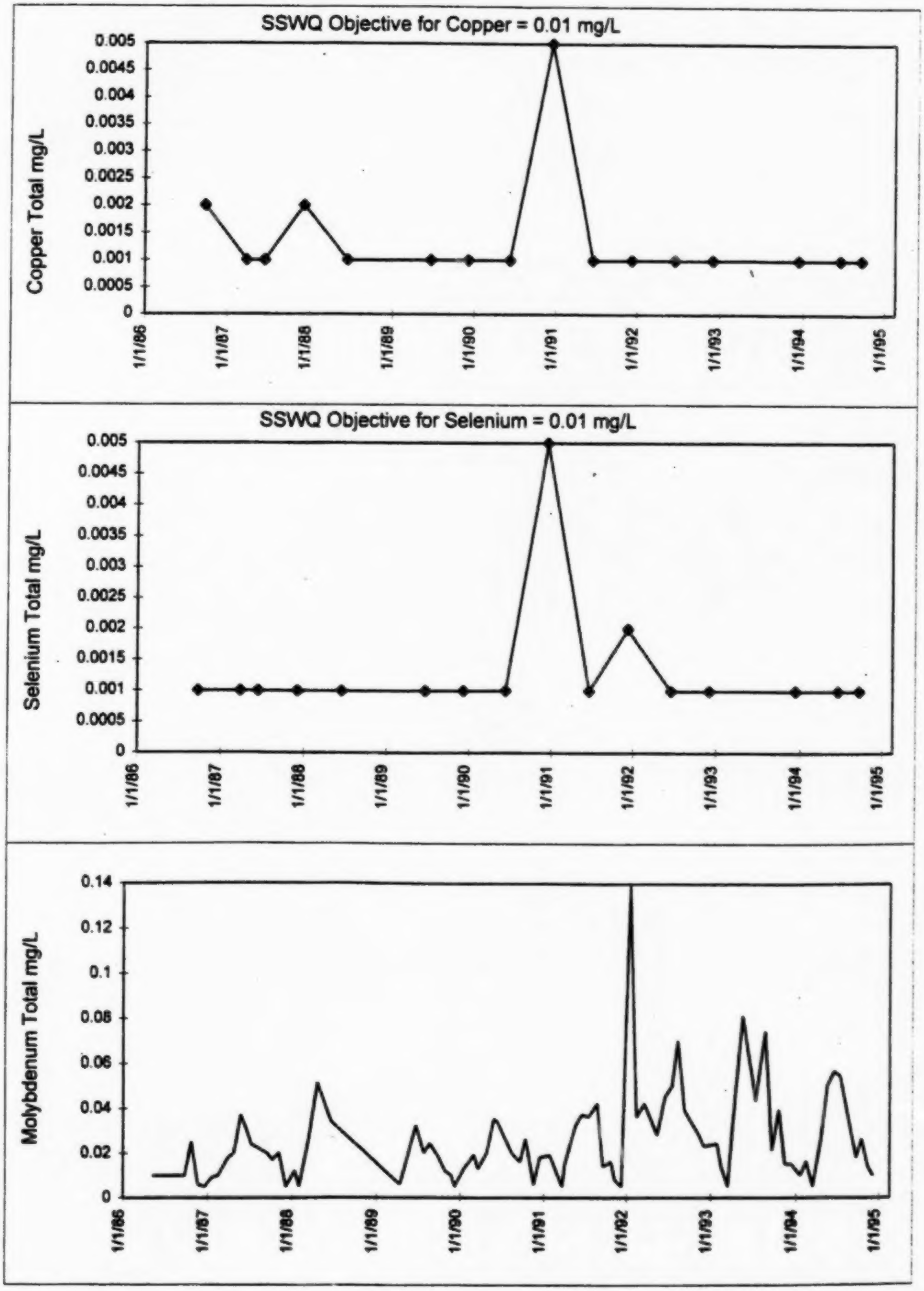
Parameter	Objectives
Bacteria:	
Total Coliforms	Not to exceed a density of 5,000 per 100 mL in any sample.
Fecal Coliform	Not to exceed a density of 1,000 per 100 mL in any sample.
Biochemical Oxygen Demand (BOD_5)	Dependent on the assimilative capacity of the receiving water. The BOD must not exceed a limit which would create a dissolved oxygen content of less than five mg/L.
Colour (apparent)	Not to be increased by more than 30 colour units above ambient natural value.
Flow	Water quantities (flow and lake levels) should not be altered unreasonably so as to result in water quality impairment such that water uses are adversely affected.
Major Ions	The predominant cations of sodium, calcium, and magnesium, and anions of sulphate, chloride and bicarbonate are too variable in the natural water quality state to attempt to define objectives. In general, however, levels of these major ions should not be suddenly increased or decreased in a receiving water so as to result in injury to aquatic life, or alter the sodium:cation ratio beyond the range of 0.30 to 0.75.
Nutrients	Nitrogen or phosphorus or other nutrient concentrations should not be altered from natural levels by discharges of effluents such that nuisance growths of algae or aquatic weeds result.
Oxygen (dissolved)	A minimum of five mg/L at any time.
Pesticides	To provide reasonably safe concentrations of these materials in receiving waters an application shall not exceed 1/100 of the 96 hr LC_{50}. Consideration must also be given to sublethal effects and to objectionable taste and odour generation, and to tainting of fish flesh.
pH	To be in the range of 6.5 to 8.5 pH units but not altered by more than 0.5 pH units from ambient values.
Radium 226	0.11 Bq/L
Suspended Solids	Not to be increased by more than 10 mg/L over existing background values for waters with levels of less than 100 mg/L, or not to be increased by more than 10 percent over existing values for waters with levels greater than 100 mg/L.
Sulphide	0.01 mg/L
Turbidity	Not to be increased by more than 25 turbidity units above ambient value.

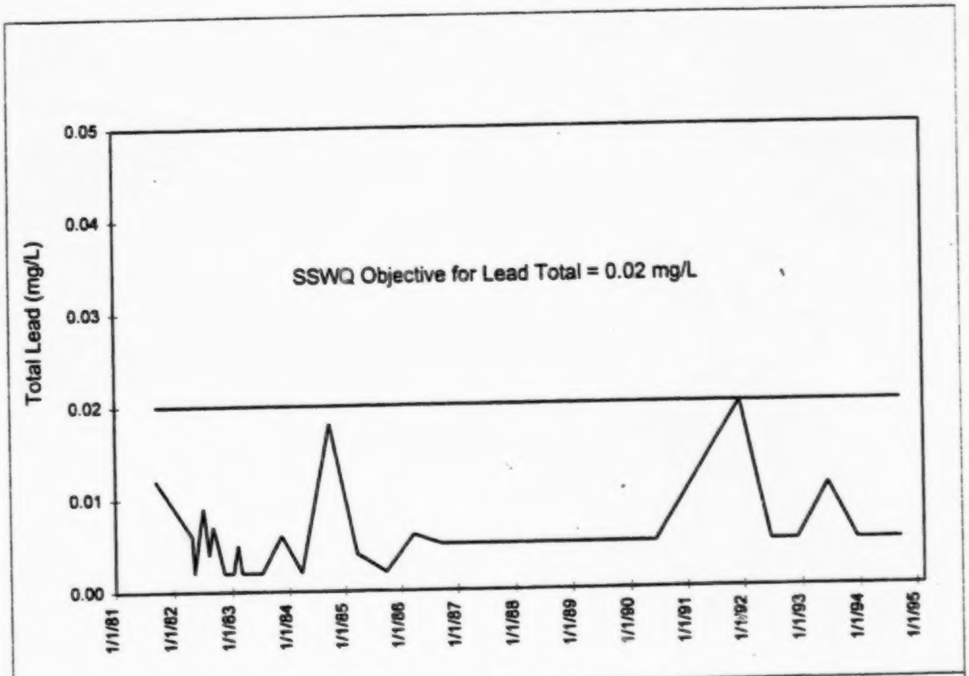
Appendix 5: Water Quality Data for Island Creek at the Dolomites

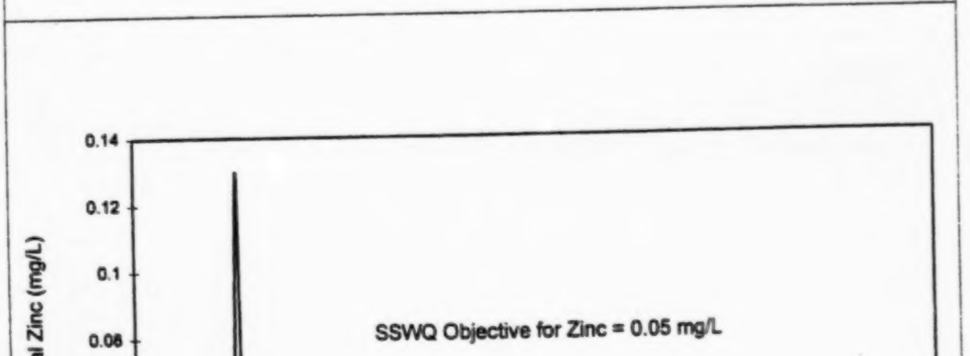


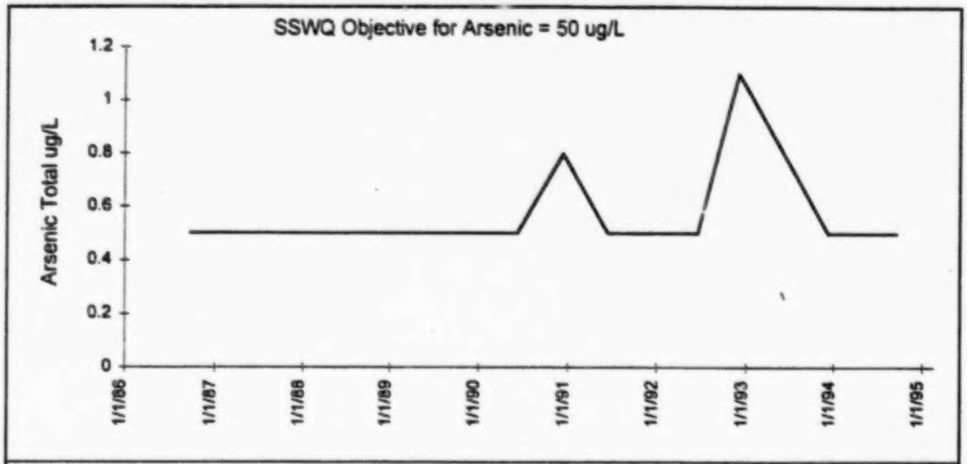
SSWQ Objective for Vanadium (Livestock Protection) = 0.1mg/L



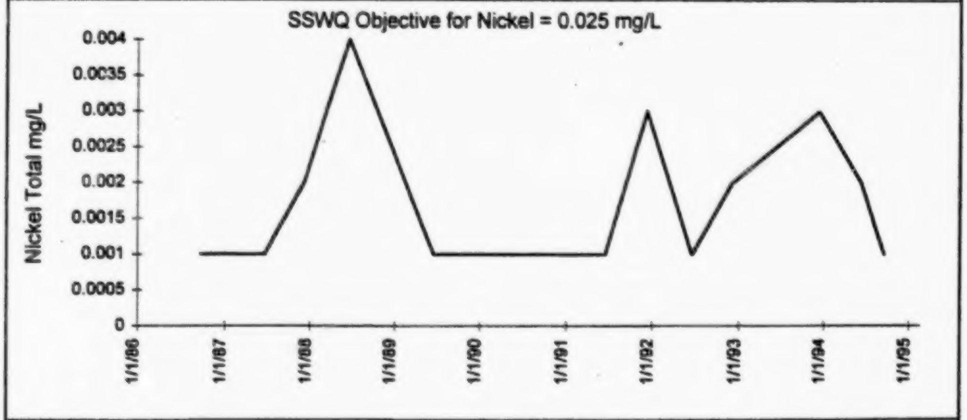

SSWQ Objective for Copper = 0.01 mg/L
Copper Total mg/L
0.005
0.0045
0.004
0.0035
0.003
0.0025
0.002
0.0015
0.001
0.0005
0
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
SSWQ Objective for Selenium = 0.01 mg/L
Selenium Total mg/L
0.005
0.0045
0.004
0.0035
0.003
0.0025
0.002
0.0015
0.001
0.0005
0
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
Molybdenum Total mg/L
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95


Total Lead (mg/L)
0.05
0.04
0.03
0.02
0.01
0.00
SSWQ Objective for Lead Total = 0.02 mg/L
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95

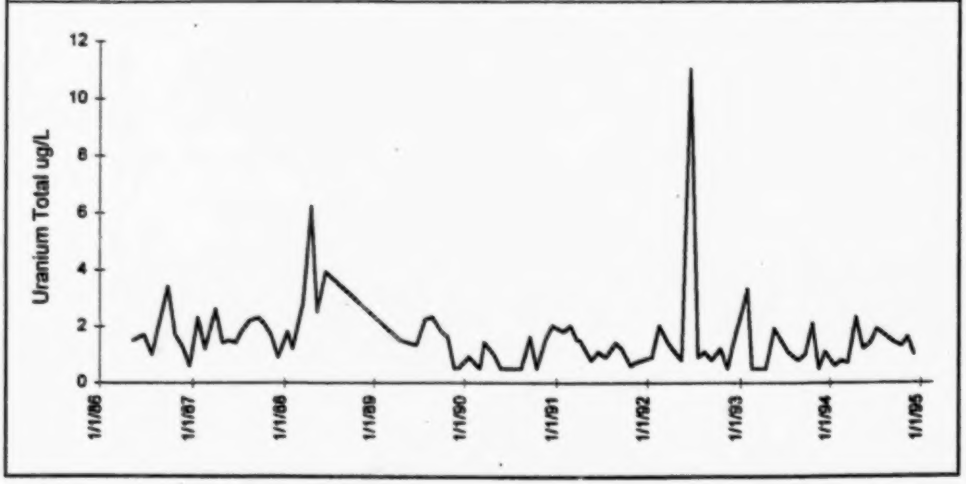

Total Zinc (mg/L)
0.14
0.12
0.1
0.08
0.06
0.04
0.02
0
SSWQ Objective for Zinc = 0.05 mg/L
1/1/81
1/1/82
1/1/83
1/1/84
1/1/85
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95

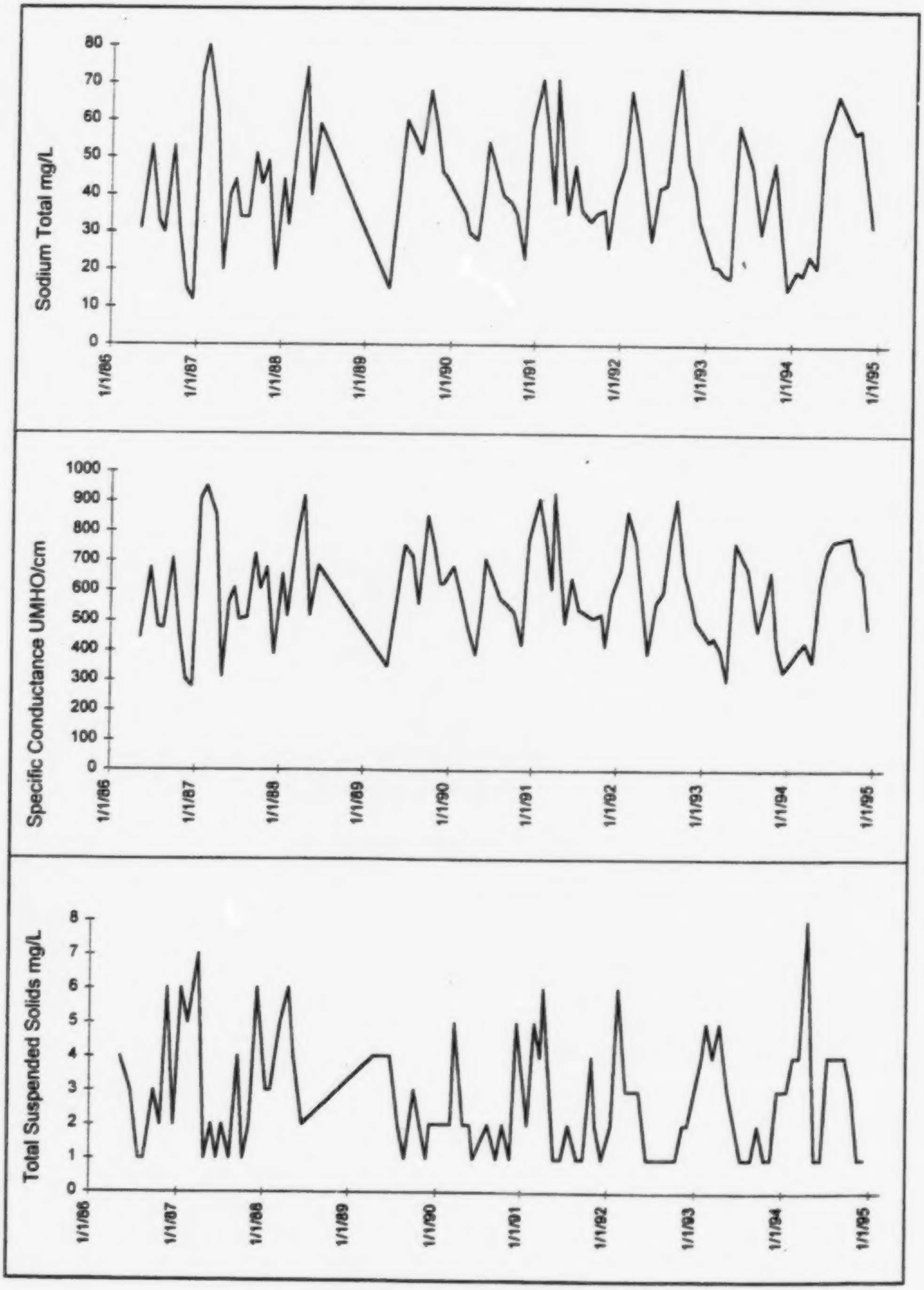


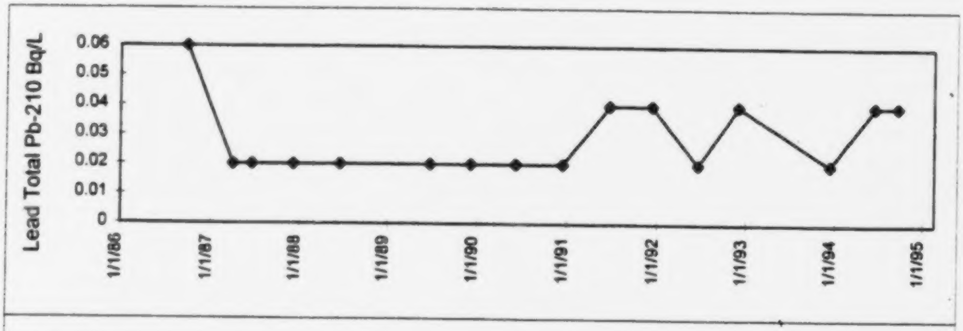
SSWQ Objective for Arsenic = 50 ug/L

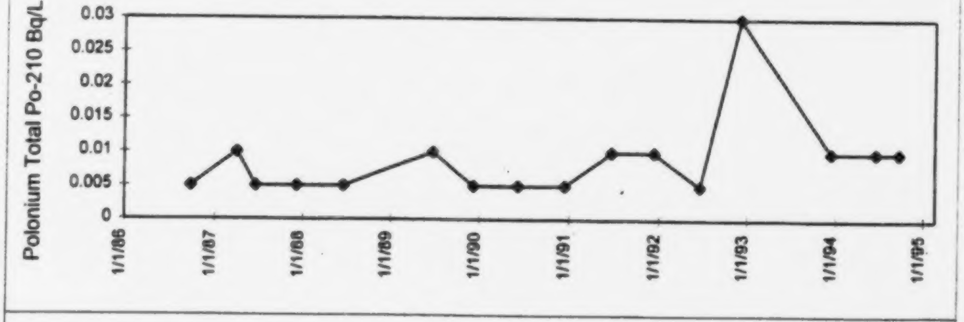


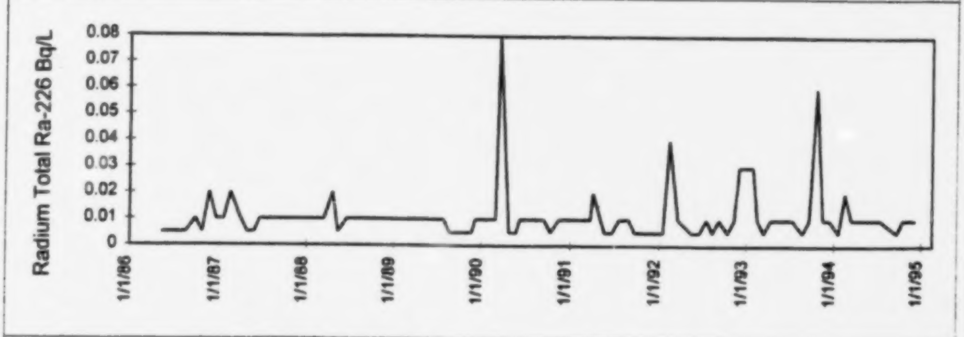
SSWQ Objective for Nickel = 0.025 mg/L

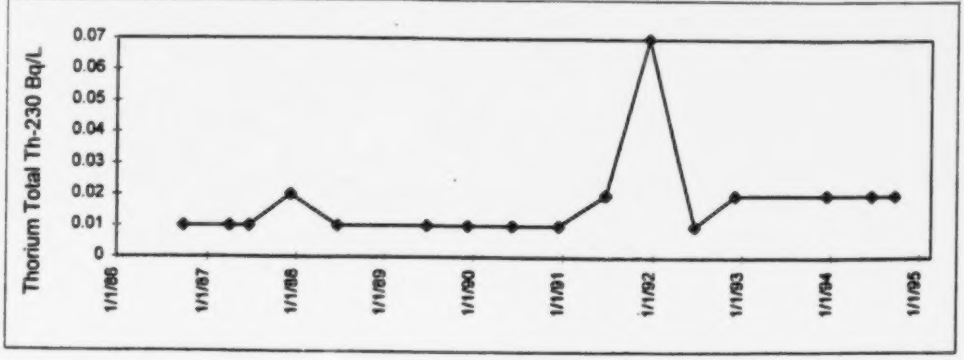


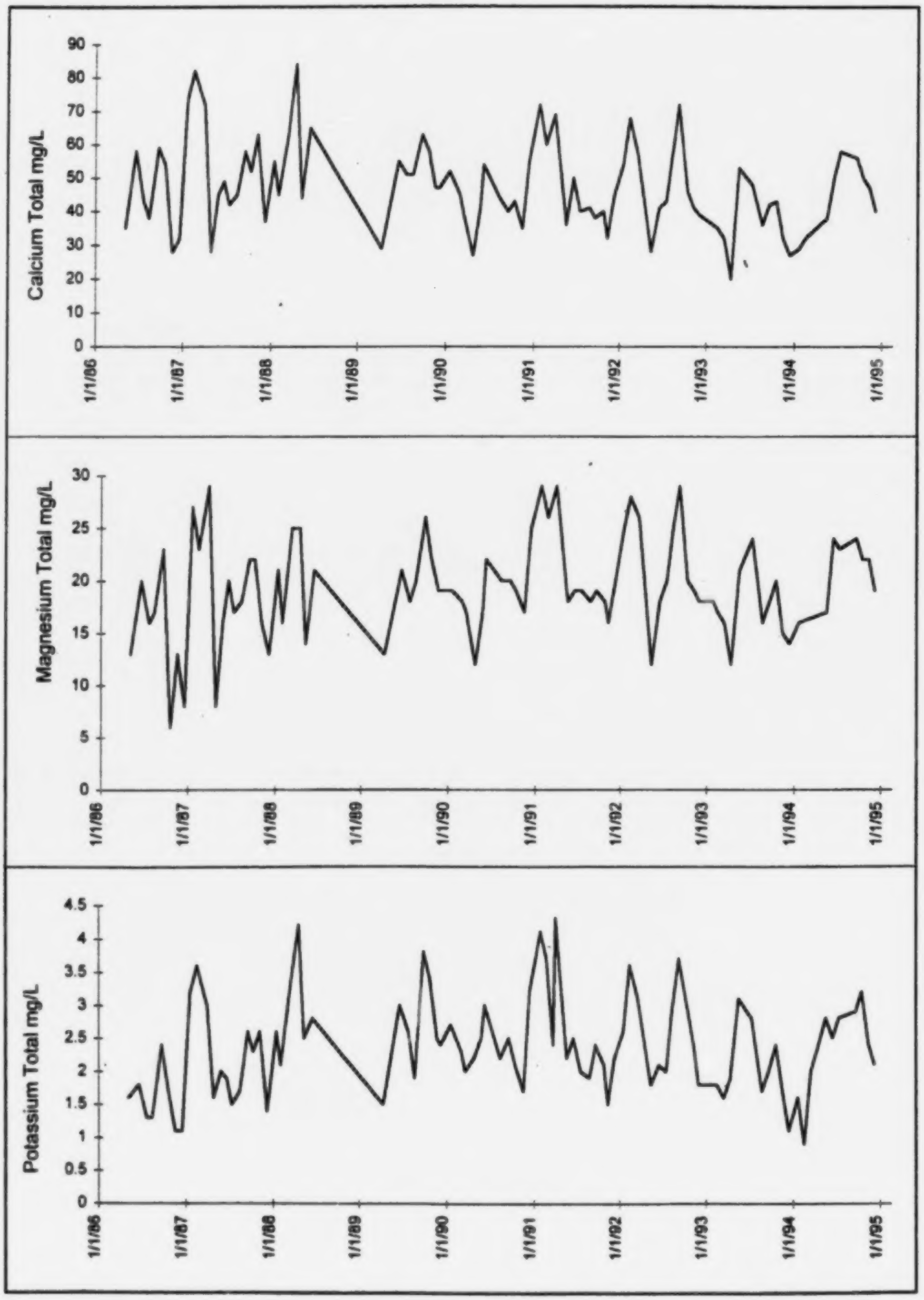

Sodium Total mg/L
0
10
20
30
40
50
60
70
80
Specific Conductance UMHO/cm
0
100
200
300
400
500
600
700
800
900
1000
Total Suspended Solids mg/L
0
1
2
3
4
5
6
7
8
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95


Lead Total Pb-210 Bq/L
0.06
0.05
0.04
0.03
0.02
0.01
0
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95


Polonium Total Po-210 Bq/L
0.03
0.025
0.02
0.015
0.01
0.005
0
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95


Radium Total Ra-226 Bq/L
0.08
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95


Thorium Total Th-230 Bq/L
0.07
0.06
0.05
0.04
0.03
0.02
0.01
0
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95


Calcium Total mg/L
90
80
70
60
50
40
30
20
10
0
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
Magnesium Total mg/L
30
25
20
15
10
5
0
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95
Potassium Total mg/L
4.5
4
3.5
3
2.5
2
1.5
1
0.5
0
1/1/86
1/1/87
1/1/88
1/1/89
1/1/90
1/1/91
1/1/92
1/1/93
1/1/94
1/1/95

Appendix 6: Douglas River at Sandy Lake: Outlet vs. Inlet Water Quality Comparison

Analyte	Units	SWQO	Statistics	1989	1990	1991	1992	1993	1994	Overall
pH		6.5-8.5	Mean (Outlet)	7.11	7.42	7.24	7.34	7.21	7.37	-
			Mean diff.[1]	0.05	0.08	0.12	0.07	0.08	0.14	0.08
			N[2]	4	4	4	4	3	2	21
			Sig. Changes[3]	-	-	-	-	-	Increase	Increase
Sp. Cond.	umhos •cm^{-1}		Mean (Outlet)	172	178	172	170	206	193	-
			Mean diff.	18	5	36	1	-12	26	12
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	Increase	-	-	-	Increase
TSS	mg•L^{-1}		Mean (Outlet)	2.3	2.8	1	1	1.3	1.5	-
			Mean diff.	-0.8	-1.8	-0.8	-0.5	-0.3	-1	-0.9
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	-	-	-	Decrease	Decrease
Turbidity	(NTU)		Mean (Outlet)	1	0.7	0.9	0.8	3.8	0.7	-
			Mean diff.	-0.1	-0.2	-0.2	-0.1	3	-0.2	0.3
			N	4	4	4	4	3	2	21
			Sig. Changes	-	Decrease	-	-	-	-	
TDS	mg•L^{-1}		Mean (Outlet)	116	127	106	129	147	124	-
			Mean diff.	4	-5	12	2	-6	15	3
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	-	-	-	-	-
Total Alkalin.	mg•L^{-1}		Mean (Outlet)	36	36	35	36	35	35	-
			Mean diff.	4	6	7	4	3	7	5
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	Increase	-	-	-	Increase
Total Hard.	mg•L^{-1}		Mean (Outlet)	65	72	68	72	83	82	-
			Mean diff.	5	3	8	1	0	17	5
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	-	-	-	-	Increase
CO_3	mg•L^{-1}		Mean (Outlet)	-	-	-	-	1	1	-
			Mean diff.	-	-	-	-	0	0	0
			N	-	-	-	-	3	2	5
			Sig. Changes	-	-	-	-	-	-	-
HCO_3	mg•L^{-1}		Mean (Outlet)	43	44	43	43	43	42	-
			Mean diff.	4	8	8	5	4	8	6
			N	4	4	4	4	3	2	21
			Sig. Changes	Increase	Increase	Increase	-	-	-	Increase
SO_4	mg•L^{-1}		Mean (Outlet)	3	3.3	3.5	4.3	3.3	5.7	-
			Mean diff.	2	2.1	2.3	2.7	1.9	4.6	2.4
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	-	Increase	-	-	Increase

Analyte	Units	SWQO	Statistics	1989	1990	1991	1992	1993	1994	Overall
Cl	mg•L⁻¹		Mean (Outlet)	28	29	25	24	33	29	-
			Mean diff.	1	-3	4	-5	-6	1	-1
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	Increase	Decrease	-	-	-
Ca	mg•L⁻¹		Mean (Outlet)	16	16	16	15	18	16	-
			Mean diff.	1	1	3	0	0	3	1
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	Increase	-	-	-	Increase
Mg	mg•L⁻¹		Mean (Outlet)	7	8	7	8	9	10	-
			Mean diff.	1	0	0	0	0	3	0
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	-	-	-	-	-
Na	mg•L⁻¹		Mean (Outlet)	4	4	4	4	5	6	-
			Mean diff.	1	0	1	0	0	1	1
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	-	-	-	-	Increase
Sums of Ions	mg•L⁻¹		Mean (Outlet)	102	105	99	100	113	109	-
			Mean diff.	10	9	18	4	-1	20	9
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	Increase	-	-	-	Increase
K	mg•L⁻¹		Mean (Outlet)	1.1	1	1	0.9	1.1	0.8	-
			Mean diff.	0.1	0	0.1	0	-0.1	-0.1	0
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	-	-	-	-	-
N-total	mg•L⁻¹		Mean (Outlet)	0.3	0.35	0.51	0.16	0.25	0.22	-
			Mean diff.	0.05	-0.07	0.16	0.01	0.06	0.04	0.04
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
N-Ammonia	mg•L⁻¹	Variable	Mean (Outlet)	0.02	0.11	0.01	0.03	0.02	0.02	-
			Mean diff.	0	0.08	0	-0.01	-0.02	0	0.01
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
N-NO_2+NO_3	mg•L⁻¹		Mean (Outlet)	0.01	0.05	0.03	0.01	0.03	0.01	-
			Mean diff.	0	0.02	0	-0.04	0	0	0
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
N-Kjeldahl	mg•L⁻¹		Mean (Outlet)	0.3	0.3	0.49	0.13	0.23	0.22	-
			Mean diff.	0.05	-0.08	0.16	0.01	0.06	0.04	0.04
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-

Analyte	Units	SWQO	Statistics	1989	1990	1991	1992	1993	1994	Overall
P-total	mg•L^{-1}		Mean (Outlet)	0.02	0.02	0.04	0.01	0.01	0.02	-
			Mean diff.	-0.03	-0.12	0.03	0	0	-0.01	-0.02
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
Al	mg•L^{-1}		Mean (Outlet)	0.027	0.032	0.029	0.042	0.03	0.919	-
			Mean diff.	-0.007	-0.006	-0.009	-0.012	-0.077	-0.603	-0.075
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	-	Decrease	-	-	-
As	ug•L^{-1}	50.0	Mean (Outlet)	0.5	0.5	0.5	1.7	0.6	0.5	-
			Mean diff.	-0.1	0	0	0.7	0.1	0	0.1
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
B	mg•L^{-1}		Mean (Outlet)	0.08	0.05	0.05	0.05	0.05	0.05	-
			Mean diff.	0	0	0	0	0	0	0
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
Ba	mg•L^{-1}	1.0	Mean (Outlet)	0.009	0.013	0.011	0.015	0.013	0.014	-
			Mean diff.	0	0.002	0	-0.002	-0.003	0	-0.001
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
Bi	ug•L^{-1}		Mean (Outlet)	5	5	5	5	1	1	-
			Mean diff.	0	0	0	0	0	0	0
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
Cd	mg•L^{-1}	0.001	Mean (Outlet)	0.001	0.001	0.001	0.001	0.001	0.001	-
			Mean diff.	0	0	0	0	0	0	0
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
Co	mg•L^{-1}		Mean (Outlet)	0.001	0.001	0.0015	0.001	0.001	0.001	
			Mean diff.	0	0	0	0	0	0	-
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
Cr	mg•L^{-1}	0.020	Mean (Outlet)	0.003	0.001	0.001	0.008	0.009	0.002	-
			Mean diff.	-0.002	-0.001	0	-0.002	0.001	0.001	-0.001
			N	2	2	2	2	2	2	12
			Sig. Changes	-	Decrease	-	-	-	-	-
Cu	mg•L^{-1}	0.01	Mean (Outlet)	0.001	0.001	0.001	0.002	0.001	0.002	-
			Mean diff.	0	0	0	0	0	0.001	0
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-

Analyte	Units	SWQO	Statistics	1989	1990	1991	1992	1993	1994	Overall
Fe	ug•L^{-1}	1.0	Mean (Outlet)	0.35	0.49	0.5	0.51	0.455	0.38	-
			Mean diff.	-0.135	0.045	-0.165	-0.095	-0.175	-0.115	-0.107
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	Decrease
Hg	ug•L^{-1}	0.1	Mean (Outlet)	0.05	0.05	0.05	0.05	0.08	0.09	-
			Mean diff.	0	0	0	0	0.01	0.04	0.01
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
Mn	ug•L^{-1}		Mean (Outlet)	0.011	0.013	0.012	0.015	0.016	0.012	-
			Mean diff.	-0.002	0	-0.003	-0.005	-0.022	-0.003	-0.006
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
Mo	ug•L^{-1}		Mean (Outlet)	0.005	0.006	0.006	0.008	0.009	0.005	-
			Mean diff.	0	0.001	0.001	0.003	0.002	0	0.001
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	-	-	-	-	-
Ni	ug•L^{-1}	0.025	Mean (Outlet)	0.001	0.001	0.001	0.003	0.002	0.002	-
			Mean diff.	0	0	0	0	-0.004	0.001	-0.001
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
Pb	ug•L^{-1}	0.02	Mean (Outlet)	0.007	0.005	0.005	0.009	0.005	0.005	-
			Mean diff.	0.002	0	0	0.002	0	0	0.001
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
Se	ug•L^{-1}	0.01	Mean (Outlet)	0.001	0.004	0.002	0.001	0.001	0.001	-
			Mean diff.	0	0.001	-0.001	0	0	0	0
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
Zn	ug•L^{-1}	0.05	Mean (Outlet)	0.008	0.005	0.005	0.007	0.005	0.036	-
			Mean diff.	-0.008	0	0	0	0	0.003	-0.001
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
^{210}Pb	Bq•L^{-1}		Mean (Outlet)	0.045	0.02	0.04	0.02	0.04	0.04	-
			Mean diff.	0.02	0	0	0	0	0	0.003
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
^{210}Po	Bq•L^{-1}		Mean (Outlet)	0.005	0.005	0.01	0.008	0.01	0.01	-
			Mean diff.	0	-0.001	0	0.003	0	0	0
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-

Analyte	Units	SWQO	Statistics	1989	1990	1991	1992	1993	1994	Overall
^{226}Ra	Bq•L^{-1}	0.11	Mean (Outlet)	0.008	0.008	0.006	0.01	0.008	0.008	-
			Mean diff.	-0.005	-0.001	-0.004	0.001	0	0	-0.002
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	-	-	-	-	-
^{230}Th	Bq•L^{-1}		Mean (Outlet)	0.01	0.01	0.02	0.01	0.02	0.02	-
			Mean diff.	0	-0.005	0	0	-0.01	-0.005	-0.003
			N	2	2	2	2	2	2	12
			Sig. Changes	-	-	-	-	-	-	-
U	ug•L^{-1}		Mean (Outlet)	0.825	0.6	0.5	0.5	0.967	0.5	
			Mean diff.	0.125	0.1	0	-0.95	-0.2	0	-0.167
			N	4	4	4	4	3	2	21
			Sig. Changes	-	-	-	-	-	-	-

[1] Mean of differences between outlet and inlet stations, where difference = outlet concentration - inlet concentration.

[2] Number of samples measured at both outlet and inlet stations on the same date.

[3] Significant increase/decrease concluded from t test with 0.05 significance level.

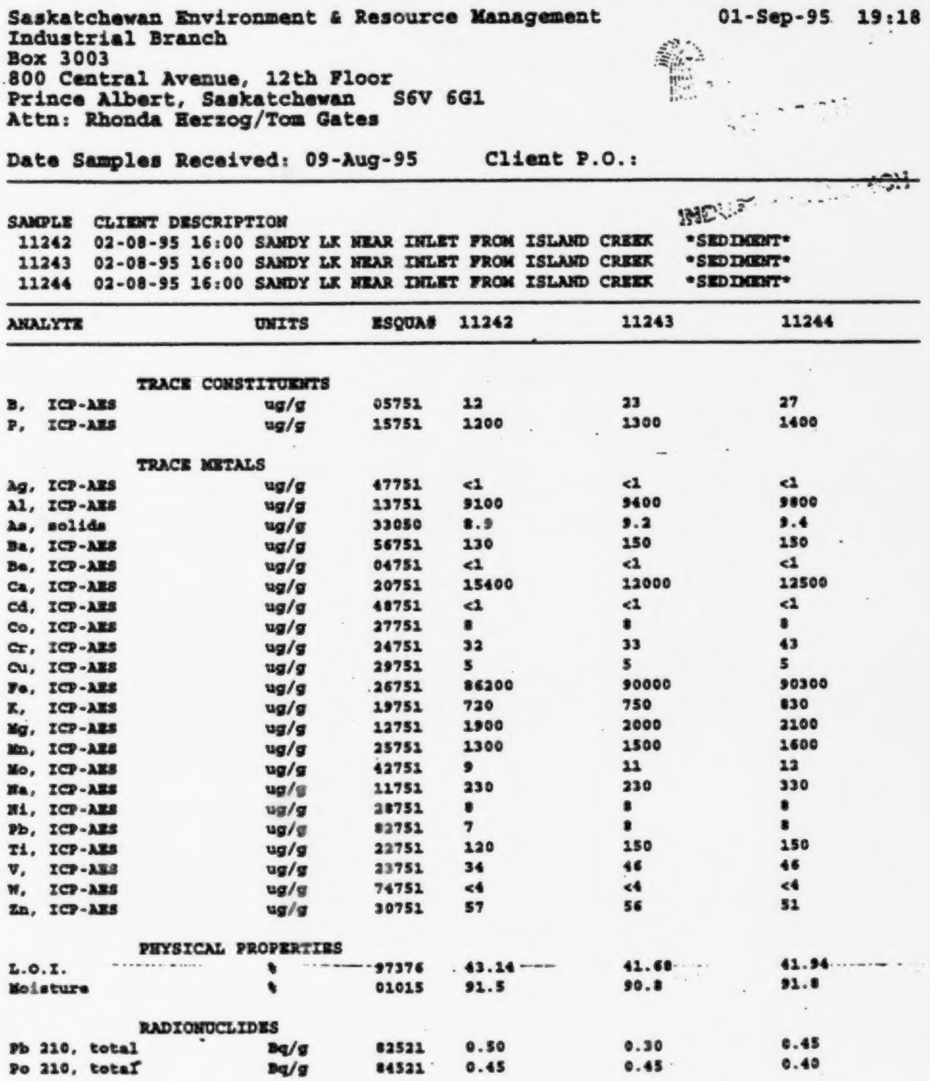
Appendix 7: Draft Interim Freshwater Sediment Guidelines for Metals

Threshold effect levels (TELS), probable effect levels (PELS), and incidence (%) of adverse biological effects in concentration ranges defined by these values (Smith *et al.* (1995) The Development and Implementation of Canadian Sediment Quality Objectives).

Metal	TEL	PEL	% ≤ TEL	TEL < % < PEL	% ≥ PEL
Arsenic	5.90	17.0	5	25	12
Cadmium	0.596	3.53	11	12	47
Chromium	37.3	90.0	2	19	49
Copper	35.7	197	4	38	44
Lead	35.0	91.3	5	23	42
Mercury	0.174	0.486	8	34	36
Nickel	18.0	35.9	4	18	44
Zinc	123	315	5	32	36

Draft TELs and PELs presented here have been calculated based on the information compiled as of January 1994.

Appendix 8: SERM CEM Sediment Quality Data for Sandy Lake and Control Site



Saskatchewan Environment & Resource Management 01-Sep-95 19:18
Industrial Branch
Box 3003
800 Central Avenue, 12th Floor
Prince Albert, Saskatchewan S6V 6G1
Attn: Rhonda Herzog/Tom Gates

Date Samples Received: 09-Aug-95 Client P.O.:

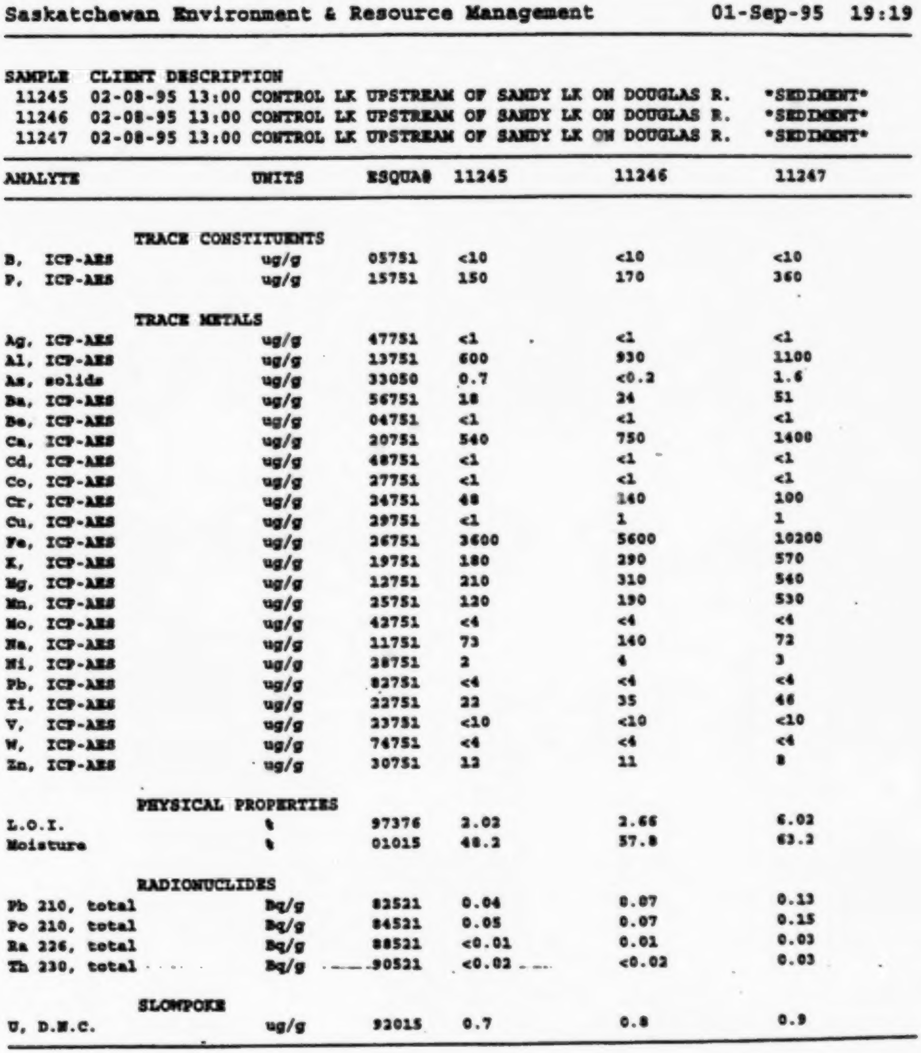
SAMPLE	CLIENT DESCRIPTION	
11242	02-08-95 16:00 SANDY LK NEAR INLET FROM ISLAND CREEK	*SEDIMENT*
11243	02-08-95 16:00 SANDY LK NEAR INLET FROM ISLAND CREEK	*SEDIMENT*
11244	02-08-95 16:00 SANDY LK NEAR INLET FROM ISLAND CREEK	*SEDIMENT*

ANALYTE	UNITS	ESQUA#	11242	11243	11244
TRACE CONSTITUENTS					
B, ICP-AES	ug/g	05751	12	23	27
P, ICP-AES	ug/g	15751	1200	1300	1400
TRACE METALS					
Ag, ICP-AES	ug/g	47751	<1	<1	<1
Al, ICP-AES	ug/g	13751	9100	9400	9800
As, solids	ug/g	33050	8.9	9.2	9.4
Ba, ICP-AES	ug/g	56751	130	150	150
Be, ICP-AES	ug/g	04751	<1	<1	<1
Ca, ICP-AES	ug/g	20751	15400	12000	12500
Cd, ICP-AES	ug/g	48751	<1	<1	<1
Co, ICP-AES	ug/g	27751	8	8	8
Cr, ICP-AES	ug/g	24751	32	33	43
Cu, ICP-AES	ug/g	29751	5	5	5
Fe, ICP-AES	ug/g	26751	86200	90000	90300
K, ICP-AES	ug/g	19751	720	750	830
Mg, ICP-AES	ug/g	12751	1900	2000	2100
Mn, ICP-AES	ug/g	25751	1300	1500	1600
Mo, ICP-AES	ug/g	42751	9	11	12
Na, ICP-AES	ug/g	11751	230	230	330
Ni, ICP-AES	ug/g	28751	8	8	8
Pb, ICP-AES	ug/g	82751	7	8	8
Ti, ICP-AES	ug/g	22751	120	150	150
V, ICP-AES	ug/g	23751	34	46	46
W, ICP-AES	ug/g	74751	<4	<4	<4
Zn, ICP-AES	ug/g	30751	57	56	51
PHYSICAL PROPERTIES					
L.O.I.	%	97376	43.14	41.68	41.94
Moisture	%	01015	91.5	90.8	91.8
RADIONUCLIDES					
Pb 210, total	Bq/g	82521	0.50	0.30	0.45
Po 210, total	Bq/g	84521	0.45	0.45	0.40

Saskatchewan Environment & Resource Management **01-Sep-95 19:19**

ANALYTE	UNITS	ESQUA#	11242	11243	11244
Ra 226, total	Bq/g	88521	0.40	0.40	0.40
Th 230, total	Bq/g	90521	0.03	0.03	0.03
SLOWPOKE					
U, D.N.C.	ug/g	93015	10.8	10.3	10.9

All results are reported on a dry basis.



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Saskatchewan Environment & Resource Management          01-Sep-95  19:19
_________________________________________________________________________

SAMPLE  CLIENT DESCRIPTION
 11245  02-08-95 13:00 CONTROL LK UPSTREAM OF SANDY LK ON DOUGLAS R.   *SEDIMENT*
 11246  02-08-95 13:00 CONTROL LK UPSTREAM OF SANDY LK ON DOUGLAS R.   *SEDIMENT*
 11247  02-08-95 13:00 CONTROL LK UPSTREAM OF SANDY LK ON DOUGLAS R.   *SEDIMENT*
_________________________________________________________________________
ANALYTE                  UNITS     ESQUA#  11245        11246        11247
_________________________________________________________________________

             TRACE CONSTITUENTS
B,  ICP-AES               ug/g     05751   <10          <10          <10
P,  ICP-AES               ug/g     15751   150          170          360

             TRACE METALS
Ag, ICP-AES               ug/g     47751   <1           <1           <1
Al, ICP-AES               ug/g     13751   600          930          1100
As, solids                ug/g     33050   0.7          <0.2         1.6
Ba, ICP-AES               ug/g     56751   18           24           51
Be, ICP-AES               ug/g     04751   <1           <1           <1
Ca, ICP-AES               ug/g     20751   540          750          1400
Cd, ICP-AES               ug/g     48751   <1           <1           <1
Co, ICP-AES               ug/g     27751   <1           <1           <1
Cr, ICP-AES               ug/g     24751   48           140          100
Cu, ICP-AES               ug/g     29751   <1           1            1
Fe, ICP-AES               ug/g     26751   3600         5600         10200
K,  ICP-AES               ug/g     19751   180          290          570
Mg, ICP-AES               ug/g     12751   210          310          540
Mn, ICP-AES               ug/g     25751   120          190          530
Mo, ICP-AES               ug/g     42751   <4           <4           <4
Na, ICP-AES               ug/g     11751   73           140          72
Ni, ICP-AES               ug/g     28751   2            4            3
Pb, ICP-AES               ug/g     82751   <4           <4           <4
Ti, ICP-AES               ug/g     22751   22           35           46
V,  ICP-AES               ug/g     23751   <10          <10          <10
W,  ICP-AES               ug/g     74751   <4           <4           <4
Zn, ICP-AES               ug/g     30751   12           11           8

             PHYSICAL PROPERTIES
L.O.I.                    %        97376   2.02         2.66         6.02
Moisture                  %        01015   48.2         57.8         63.2

             RADIONUCLIDES
Pb 210, total             Bq/g     82521   0.04         0.07         0.13
Po 210, total             Bq/g     84521   0.05         0.07         0.15
Ra 226, total             Bq/g     88521   <0.01        0.01         0.03
Th 230, total             Bq/g     90521   <0.02        <0.02        0.03

             SLOWPOKE
U, D.N.C.                 ug/g     92015   0.7          0.8          0.9
_________________________________________________________________________
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Saskatchewan Environment & Resource Management 01-Sep-95 19:19

SAMPLE	CLIENT DESCRIPTION
11248	02-08-95 17:30 SANDY LAKE INLET FROM DOUGLAS RIVER *SEDIMENT*
11249	02-08-95 17:30 SANDY LAKE INLET FROM DOUGLAS RIVER *SEDIMENT*
11250	02-08-95 17:30 SANDY LAKE INLET FROM DOUGLAS RIVER *SEDIMENT*

ANALYTE	UNITS	ESQUA#	11248	11249	11250
TRACE CONSTITUENTS					
B, ICP-AES	ug/g	05751	21	21	<10
P, ICP-AES	ug/g	15751	1100	1000	1200
TRACE METALS					
Ag, ICP-AES	ug/g	47751	<1	<1	<1
Al, ICP-AES	ug/g	13751	9400	9600	9500
As, solids	ug/g	33050	6.2	5.9	5.9
Ba, ICP-AES	ug/g	56751	100	98	110
Be, ICP-AES	ug/g	04751	<1	<1	<1
Ca, ICP-AES	ug/g	20751	7500	6700	7200
Cd, ICP-AES	ug/g	48751	<1	<1	<1
Co, ICP-AES	ug/g	27751	6	6	6
Cr, ICP-AES	ug/g	24751	67	42	71
Cu, ICP-AES	ug/g	29751	6	5	5
Fe, ICP-AES	ug/g	26751	50000	47100	50900
K, ICP-AES	ug/g	19751	960	1000	810
Mg, ICP-AES	ug/g	12751	1600	1500	1500
Mn, ICP-AES	ug/g	25751	680	610	710
Mo, ICP-AES	ug/g	42751	<4	<4	<4
Na, ICP-AES	ug/g	11751	220	160	160
Ni, ICP-AES	ug/g	28751	7	6	7
Pb, ICP-AES	ug/g	82751	5	5	5
Ti, ICP-AES	ug/g	22751	130	160	130
V, ICP-AES	ug/g	23751	35	33	34
W, ICP-AES	ug/g	74751	<4	<4	<4
Zn, ICP-AES	ug/g	30751	53	38	39
PHYSICAL PROPERTIES					
L.O.I.	%	97376	37.07	35.36	37.46
Moisture	%	01015	88.1	87.5	88.2
RADIONUCLIDES					
Pb 210, total	Bq/g	82521	0.30	0.25	0.30
Po 210, total	Bq/g	84521	0.30	0.25	0.30
Ra 226, total	Bq/g	88521	0.30	0.30	0.30
Th 230, total	Bq/g	90521	0.03	0.02	0.03
SLOWPOKE					
U, D.N.C.	ug/g	92015	5.0	4.6	5.0